Environmental change and high status water-bodies


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I confirm that the word count of this thesis is less than 100,000 words.

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List of abbreviations

ASPT	Average Score Per Taxon
BOD	Biochemical Oxygen Demand
BMWP	Biological Monitoring Working Party
CSI	Channel Substrate Index
CLC	CORINE Land Cover
CoFSI	Combined Fine Sediment Index
CORINE	Coordination of Information on the Environment
CSAs	Critical Source Areas
DAFM	Department of Agriculture, Food and the Marine
DO	Dissolved Oxygen
DEHLI	Drought Effect of Habitat Loss on Invertebrates
DJFM	December, January, February and March (Winter Months)
EQR	Ecological Quality Ratio
ESB	Electrical Supply Board
E-PSI	Empirically-Weighted PSI
EIA	Environmental Impact Assessment
EPA	Environmental Protection Agency
EPT	Ephemeroptera, Plecoptera and Trichoptera
MS	EU Member States
EU	European Union
FSBI	Fine Sediment Biotic Index
GIS	Geographic Information System
GLAS	Green Low-Carbon Agri-Environmental Scheme
HMWBs	Heavily Modified Water-bodies
HSWs	High Status Water-Bodies
HaRMONy	High Status Waterbodies: Managing and Optimising Nutrients
INNS	Invasive Non-Native Species

LIFE	Lotic Index for Flow Evaluation
Nitrogen	N
NGP	Nationaler Gewässerbewirtschaftungsplan
GBNIIENB	Neagh-Bann RBD
NAO	North Atlantic Oscillation
GBNIIENW	North Western RBD
NTAXA	Number of Scoring Taxa
OPW	Office of Public Works
oFSI	Organic Fine Sediment Index
P	Phosphorus
PRP	Pollution Reduction Programmes
PES	Polyethersulfone
POM	Programme of Measures
PSI	Proportion of Sediment-Sensitive Index
RICT/RIVPACS	River Invertebrate Classification Tool
RBD	River Basin District
RBMP	River Basin Management Plan
SEPA	Scottish Environmental Protection Agency
SRP	Soluble Reactive Phosphorus
JJAS	June, July, August and September (Summer Months)
SUDS	Sustainable Urban Drainage System
IESW	Ireland South Western RBD
IEWE	Ireland Western RBD
ToFSI	Total Fine Sediment Index
TON	Total Oxidized Nitrogen
EA	Environment Agency (UK)
WHPT	Whalley, Hawkes, Paisley & Trigg
WFD	Water Framework Directive
WISE	Water Information System for Europe

Preface

This PhD study forms part of the wider ***H**igh Status WateRbodies: **M**anaging and **O**ptimising **N**utrients* (HaRMONy[1] 2013-2018) project that was commissioned to investigate potential factors contributing to the large number of declines in water-bodies (rivers, lakes, transitional and coastal waters) classified as being of "high status" in Ireland. Given that the majority of high status water-bodies (HSWs) are located in upland areas, where soils are typically peaty and agriculture is practiced extensively, there is the potential for poor farm nutrient management practices to impact on sensitive waterbodies within these catchment areas. With this in mind, the HaRMONy project set out to combine research on soils and hydrology with socio-economic factors to establish management strategies that may be applied at the local level, and that serve to halt and potentially reverse the observed deteriorations in HSWs. The HaRMONy project was funded under a Department of Agriculture Food and the Marine (DAFM) Research Stimulus Fund, and was a collaboration between research scientists in Teagasc (Johnstown Castle, Co. Wexford and Athenry, Co. Galway), along with scientists from the Ulster University, NUI Galway and the Agri-Food Biosciences Institute.

Scientists associated with the HaRMONy project have an overall objective of providing "strategies for nutrient management in sensitive catchments", with this being achieved through catchment case-studies and an assessment of current nutrient management practices and farm activities in these areas. The HaRMONy project aimed to determine optimum nutrient efficiency based on the predominant soil types in these areas, as well as assess the contribution land use activities play in determining status.

[1] https://www.teagasc.ie/environment/water-quality/harmony/

As part of the HaRMONy project, this PhD study focused on the potential association between environmental change and HSW deteriorations. Furthermore, this PhD study assessed if land use and land cover change, streamflow modifications and sedimentation are factors contributing to loss of status, with these pressures serving as a counter to the nutrient impact and behavioural hypotheses, studied elsewhere by project partners.

The PhD study is presented in six chapters: Chapter 1 provides an overview of HSWs and their management strategies in the European Union; Chapter 2 assesses the relationship between land cover change and HSWs; Chapter 3 investigates hydrological (streamflow) pressures on high status rivers; Chapter 4 assesses the impact of fine sediment on high status river sites in Ireland; and Chapter 5 provides a synthesis, recommendations and conclusions based on the findings of the previous four chapters.

Chapter 1: High status water-bodies in the European Union

Chapter 2: The relationship between land cover change and high status water-bodies

Chapter 3: Investigating hydrological pressures on high status rivers

Chapter 4: Assessing the impact of fine sediment on high status river sites in Ireland

Chapter 5: Synthesis, recommendations and conclusions

Abstract

High status water-bodies (HSWs), as designated under the European Union (EU) Water Framework Directive (WFD), are rivers, lakes, transitional waters and coastal waters, that are close to natural status, representing conditions that are largely un-impacted by anthropogenic activities. These HSWs are sensitive areas that require special attention. However, in recent years large declines in the number of HSWs in Ireland have been observed, with these declines being attributed to pressures from point source pollution or unintentional discharges, along with low intensity practices potentially resulting from changes in land use and land cover. With this background, this PhD set out to present a review of HSWs and their management strategies in the European Union, and to investigate in three separate studies, the potential for HSW deteriorations to be caused by: 1) land use and land cover change; 2) hydrological (streamflow) modifications; and 3) sediment pressures. For these three studies, HSWs in Ireland were determined to have either: "Lost" their high status (e.g. gone from high to good, moderate, poor or bad); consistently "Maintained" their high status; or "Gained" in status (e.g. from good to high).

The review of HSWs in Europe (Chapter 1) highlighted how it may be counter-productive for countries to focus exclusively on achieving the "good" status objective of the WFD, while ignoring deteriorations to HSWs. Additionally, using case studies from four Member States with relatively large numbers of HSWs (Sweden, Austria, Ireland, and UK (Scotland)), the review assessed variations in strategies employed to manage HSWs. Based on these case studies it was determined that lag times between implementing management strategies and seeing actual benefits make assessing the effectiveness of such measures difficult, but that countries that have developed strategies may benefit from the sharing of knowledge, for example Ireland and Scotland.

The land cover change study (Chapter 2) demonstrated methods for assessing land cover change using CORINE data for three time periods: 2006-2012, 2000-2006 and 2000-2012; and found that anthropogenically influenced changes in land use and land cover types were linked to declines in water body status, with a higher level of natural/semi-natural land occurring in Maintained catchments. For example, in the period 2006-2012, land that changed from Forestry to Heterogeneous Agricultural areas was 17.5 times more likely to result in Lost status, whereas land that remained as Forestry or remained as Inland Wetlands reduced the chance of Lost status occurring by 15 % and 4 %, respectively. However, the similarity of land cover trends between sites that have Lost and Gained status provided further research questions. In the hydrological (streamflow) modifications study (Chapter 3), despite differences being found in Lotic Index for Flow Evaluation (LIFE) scores between the Lost and Maintained status categories, all LIFE scores were generally above 7.25 and reflective of rivers hosting invertebrate communities with a preference for medium/high streamflow rates. While some hydrometric stations in the wider study area did display changing streamflow trends, which may potentially be linked to drainage and/or change in status, the overall conclusion was that for most sites, streamflow alterations are not likely to have been a major factor leading to deteriorations. However, for certain sites, and potentially in combination with other stressors, streamflow alterations may be problematic. The sediment study (Chapter 4) found that, macro-invertebrate taxa occurring in HSWs were pre-dominantly sediment sensitive taxa.

However, for two sediment specific metrics, the Proportion of Sediment-sensitive Index (PSI) and the Empirically-weighted PSI (E-PSI), significant differences were observed between sites that Lost status and those that Maintained status, implying that at some sites, sedimentation is impacting on macro-invertebrates. Again, no difference between Lost and Gained sites was observed, leaving an important caveat. While weak to moderate relationships were observed between the sediment metrics and the physical sediment variables, no difference between status categories for any of the physical sediment variables was observed, although this may be related to the sampling resolution. Chapter 4 also highlighted the potential for multiple-stressors, such as the interaction between sediment, organic pollution and streamflow alterations, to contribute to deteriorations in status. However, nutrient sampling indicated little or no evidence of nutrient enrichment at the majority of sample sites, and it is suggested that nutrient analysis at HSWs may be better served by higher resolution monitoring. Finally, key recommendations were suggested based on the overall findings of the PhD, that included: investigating if measures being implemented in catchments with Gained status may be replicated and possibly used to improve conditions at Lost status sites; and potentially including "impacting on high status water-bodies" as an additional category requiring Environmental Impact Assessments (especially in relation to drainage works).

Chapter 1

1. High status water-bodies in the European Union

1.1. Introduction

High status water-bodies (HSWs), a European Union (EU) definition under the Water Framework Directive (WFD) (OJEC, 2000), are water-bodies (rivers, lakes, transitional waters and coastal waters) that are close to the ideal natural status, representing conditions that are largely undisturbed by anthropogenic actions (Mayes and Codling, 2009; WG 2.3, 2003). They are important because, amongst other things, they support sensitive species such as the Freshwater Pearl Mussel (*Margaritifera margaritifera*) and juvenile Salmon (*Salmo salar*), and contribute significantly to the overall species diversity of catchments (EPA, 2009; White et al., 2014). Furthermore, they benefit the public good by providing ecosystem services such as clean drinking water, as well as recreational facilities and associated economic incomes (Ní Chatháin et al., 2012). The WFD high status designation opposes bad status, depending on a condition between pristine and severely degraded, respectively, with good, moderate and poor as intermediate conditions.

The purpose of the WFD, which is the most significant piece of water resources legislation ever to have been put in place in EU member states (MS), is to provide protection for all inland surface waters, estuaries, coastal waters and groundwater bodies. Within the WFD, a further aim was for all EU water-bodies to achieve at least good ecological status or good ecological potential (for artificial or heavily modified water-bodies such as reservoirs) by 2015 (McNally, 2009; OJEC, 2000). Although significant progress has been made, a large number of water-bodies have not achieved this target and extensions, in six-year cycles, to 2027 and beyond are possible (European Commission, 2012a). Additionally, water-bodies that are already at good or high status should not deteriorate from this standing (OJEC, 2000). While the Directive does allow for exceptions to the no deterioration objective, this is only in cases where, for example: it is the result of a new sustainable development that has mitigated against adverse impacts; the reasons for the development are outlined in the River Basin Plan; there is an overriding public interest; and where no feasible alternative approach is available (OJEC, 2000).

Water pollution is a major concern for EU citizens, featuring strongly in a pole of the five greatest environmental threats, averaging at 47% of people polled (European Commission, 2015a). As high status sites represent water-bodies that in the present or past have experienced very low pressures from industrialisation, urbanisation or intensive agricultural practices, with only limited changes to their natural hydrological, chemical and biological functioning (Wallin et al., 2003; WG 2.3, 2003), it follows that they should, therefore, be especially highly regarded. However, unlike systems that are already polluted, the smallest increase in pollutants to HSWs is likely to have an impact. For example, small increases of pressures from nutrient inputs, sedimentation, flow modifications or priority substances are likely to have a disproportionate effect in comparison to already degraded sites (Ni Chatháin et al, 2012; White et al., 2014).

While one of the key objectives of the WFD is the restoration of water-bodies to good status, achieving high status is a more challenging prospect. Mao and Richards (2012) describe the difficulties associated with returning a water body to near natural status, finding little empirical evidence to demonstrate the success of restoration attempts. In contrast, Jones and Schmitz (2009) are more optimistic about the potential for restoration, but in their appraisal of 240 restoration studies they found only 20% compared restoration efforts against pre-degradation data, and only 58% used un-impacted reference sites. Although some ecological benefits are likely to be derived from restoration, they are very often not able to match those of the reference sites (Benayas et al., 2009; Bullock et al. 2011). Benayas et al. (2009) for example, found that there were substantial improvements in biodiversity and ecosystem functioning for restored sites relative to degraded sites, but a comparison between restored and reference sites yielded a median response of 86% and 80% for biodiversity and ecosystem functioning, respectively.

Emerging problems associated with land use and land cover change, invasive species and climate change further impede restoration efforts, and restoration may only be successful following a substantial time period in the region of decades or even longer (Langford et al., 2009; Whitehead et al., 2009; Hering et al., 2010; Bullock et al. 2011; Mao and Richards, 2012). This highlights the need to maintain HSWs, as any deterioration may be impossible to reverse, or at least require a large restoration time commitment. It has been suggested that the costs of restoration may be offset by the economic benefits derived from restoration (Bullock et al., 2011); however, the most cost effective approach is likely to come from halting any deteriorations or loss of high status in the first place (Ni Chatháin et al., 2012; White et al., 2014). Additionally, high status sites are refugia for sensitive species that help sustain aquatic biodiversity (Hering et al., 2010). As such they are important for the re-colonisation ability of restored water-bodies in the same system (EPA, 2009; Sundermann et al., 2011; Callanan et al., 2014), and may ultimately speed up restoration times. Sundermann et al. (2011) and Langford et al. (2009), for example, found restoration potential to be dependent on the proximity of the restored site to sources of potential colonisers and the species compositions within these pool sources.

Despite their benefits, as part of this review, a literature search of *Web of Science* with “high status” in the title and “Water Framework Directive” as a topic generated only five references. The aim of this report is, therefore, to collate data on the current standing of HSWs in the EU and to assess management strategies for their protection. The review presents case studies of four MS with relatively large numbers of water-bodies at high ecological status: Sweden, Austria, Ireland, and UK (Scotland).

1.2. Reference conditions and high status sites

When assessing high status, consideration should be given to: 1) how HSWs are determined, i.e. the relationship between high status and reference sites; and 2) the comparability of high status designations between member states. To assign the ecological status of a water body, a comparison must be drawn between the observed status of the water body (based on monitoring etc.), against an assigned “reference” condition, representing the expected status of the site if undisturbed by anthropogenic actives (Pardo et al., 2012; Stoddard et al., 2006). The resulting “ecological quality ratio” (EQR), given as a value between 0 (severely degraded or worst case) and 1 (un-impacted) is then used to classify water-bodies within the high, good, moderate, poor, and bad divisions (Birk and Hering, 2006). Bennett et al. (2011) details the process by which MS assigned reference sites, which mainly involved following the European

Commission guidance document produced by the Common Implementation Strategy Working Group 2.3 (WG 2.3, 2003). This Working Group was set up specifically to deal with the issue of establishing reference conditions and ecological status thresholds. The WFD describes reference sites as having hydrological and physico-chemical types and biological quality elements similar to those at high status sites (OJEC, 2000), while the Working Group document states that "reference conditions equal high ecological status" (WG 2.3, p. 12, 2003). While this initially led to some confusion, assigning reference sites based on abiotic criteria emerged as the solution (Phillips, 2014). Typically, however, reference sites are in the upper tier of the high status classification, and not all HSWs achieve reference level standards (McGarrigle and Lucey, 2009).

The Working Group provides a list of eight criteria for which reference sites should be screened against. These include diffuse and point source pollution, morphological modifications, water abstractions and flow alterations, bank-side vegetation types, biological concerns such as invasive species, and additional pressures such as recreational use, all of which should have negligible human influence (WG 2.3, 2003). Aligned to the establishment of reference conditions and ecological status boundaries, the WFD requires an inter-calibration approach between MS to enable comparability across country jurisdictions (Buffagni and Furse, 2006; Heiskanen et al., 2004; McGarrigle and Lucey, 2009). This inter-calibration approach aims to set EQR standards that enable a harmonised view of the expected standards for high and good ecological status throughout the EU (Heiskanen et al., 2004; Pardo et al., 2012). This has culminated in efforts to harmonise cross country methods for amongst others, macrophytes (Birk and Willby, 2010), invertebrates (Bennett et al., 2011), phytobentos (Kelly et al., 2009) and assigning reference conditions (Pardo et al., 2012).

The approach to monitoring differs across MS due in part to the differing levels and combinations of pressures present in each country. In some MS a single pressure such as organic pollution is the over-riding influence, whereas other MS are prone to several pressures of equal importance acting together (Birk and Hering, 2006; Hering et al., 2010). Birk et al. (2012) records 297 different methods being employed for biological assessment across the 28 MS, where assessment metrics are often tailored to the traditional/specific pollution concerns (Hering et al., 2010). Additionally, different approaches to setting class boundaries include the use of statistical analysis in 45% of cases, ecological methods in 37%, and expert judgement in 18% (Birk et al., 2012). Furthermore, methods for determining the level of pressures associated with reference sites include quantified measurement (34% of questionnaire respondents), field inspections (19%) and expert judgement (10%) (Pardo et al., 2012).

When assigning reference conditions, the WFD allows for methods based on either spatial (collected data) or modelled approaches, or a combination of both, or in the absence of these methods, expert judgement may be employed (OJEC, 2000). Each method has its strengths and weaknesses; for example, expert opinion may enable the incorporation of both historical and current thinking, but may also be prone to bias (WG 2.3, 2003). This may result in errors being introduced into the process of assigning reference conditions, especially as there is some ambiguity with regard to the concept of “minimally-disturbed” and the allowable level of anthropogenic pressures (Moss, 2008; Pardo et al., 2012). This in turn may result in increased type II errors, i.e. where a water body is incorrectly classified as being high status when in fact it is less than high (Hering et al., 2010). The variation in approaches itself is one source of divergence between MS when assigning reference conditions, leading to efforts at setting up standardised methods (e.g. Pardo et al., 2012). The use of non-ecological principles based on statistical distributions for assigning EQR boundaries is also questionable, as there is no guarantee linking this with any meaningful adjustments in the functioning of ecosystems or associated biological groups (Birk et al., 2012), and therefore appropriate appraisal of HSWs. Alternatively, the use of the one out all out principle, where a water body is classified based on the lowest performing ecological parameter, may result in increased type 1 errors, i.e. where a water body is classified at a lower status than is actually the case (Borja and Rodríguez 2010; Hering et al., 2010; Nõges et al., 2009; Prato et al., 2014). This method effectively allows a single element to determine the ecological status, which is contrary to the ecosystem approach proposed by the WFD (Hering et al., 2010), and may result in MS recording fewer HSWs than is actually the case.

1.3. Number of HSWs per European Member State

The number of high ecological status sites per country reported in this review is based on the "ecological and chemical status of surface water-bodies" data extracted from the Water Information System for Europe (WISE) - WFD database (EEA, 2015), which is a record of data reported by MS to WISE up to May 2012[2]. Of the 28 EU MS only Cyprus, Luxembourg and the Netherlands have no waterbodies designated as high ecological status (Table 1.1), while Malta has high status for coastal waters only. As

[2] This EEA WISE database has recently been updated (as of 20 July 2018). However, at the time of writing it was not feasible to modify this review in response to the updates.

well as the afore mentioned sites, the Czech Republic and Hungry additionally have no river sites designated as high ecological status (Tables 1.2 and 1.3), while Belgium has no lake sites designated as high ecological status (Tables 1.4 and 1.5). For many countries the status of a large percentage of water-bodies remains unknown, with for example Poland having 78% of its rivers and 82% of its lakes listed as unknown (Tables 1.2 and 1.4). The WFD sets out to manage water-bodies based on river basin parameters, as opposed to administrative or political borders (European Commission, 2015a). Member States are therefore required to draw up River Basin Management Plans (RBMPs) for defined River Basin Districts (RBDs) that set out the "programme of measures" (POM) in order to fulfil the objectives of the WFD. The first RBMPs were due in 2009, with these plans then being refreshed every six years thereafter.

However, as of 2012 only one of the 25 River Basin Districts in Spain (ES100 - Distrito Fluvial de Catalonia) had reported a RBMP to the European Commission (European Commission, 2012b); therefore, no data are recorded in the WISE-WFD data base (dated 2012) for lakes from Spain, while the rivers data are very much deficient. This has since improved to 18 RBDs reporting RBMPs for Spain, within which 10.1% of surface water-bodies are now classified as high ecological status (European Commission, 2015b), although this is not reflected in the WISE-WFD database. Similarly, Portugal and Greece and the Walloon Region in Belgium had not adopted RBMPs by 2012 (European Commission 2012a), although, unlike Spain, some data were reported to WISE. It is likely that other countries with large number of water-bodies listed as unknown, have either not yet monitored these sites or not reported data to WISE.

Table 1.1. Total number and percentage number of surface water-bodies (rivers, lakes, coastal and transitional, including HMWBs) at high, good, moderate, poor, bad and unknown ecological status for each of the 28 EU member states. Data extracted from WISE (EEA, 2015).

Country	Country	Total Count	High	High (%)	Good	Good (%)	Moderate	Moderate (%)	Poor	Poor (%)	Bad	Bad (%)	Unknown	Unknown (%)
Austria	AT	7401	1332	18	1776	24	3809	51.5	389	5.3	78	1.1	17	0.2
Belgium	BE	560	7	1.2	116	20.7	132	23.6	119	21.2	149	26.6	37	6.6
Bulgaria	BG	759	36	4.7	293	38.6	242	31.9	109	14.4	78	10.3	1	0.1
Cyprus	CY	260	8	3.1	96	36.9	82	31.5	16	6.2	4	1.5	54	20.8
Czech Republic	CZ	1140	0	0	193	16.9	155	13.6	781	68.5	0	0	11	1
Germany	DE	9863	76	0.8	912	9.2	2946	29.9	3393	34.4	2242	22.7	294	3
Denmark	DK	17984	980	5.4	6067	33.7	5275	29.3	1471	8.2	715	4	3476	19.3
Estonia	EE	750	12	1.6	522	69.6	185	24.7	28	3.7	0	0	3	0.4
Greece	EL	1575	168	10.7	517	32.8	312	19.8	175	11.1	15	1	388	24.6
Spain	ES	175	22	12.6	53	30.3	13	7.4	18	10.3	5	2.9	64	36.6
Finland	FI	6153	681	11.1	1173	19.1	849	13.8	222	3.6	58	0.9	3170	51.5
France	FR	11523	747	6.5	4024	34.9	4584	39.8	1445	12.5	468	4.1	254	2.2
Croatia	HR	1315	281	21.4	390	29.7	286	21.7	199	15.1	157	11.9	2	0.2
Hungary	HU	1082	5	0.5	100	9.2	320	29.6	193	17.8	42	3.9	422	39
Ireland	IE	5670	1012	17.8	2070	36.5	1490	26.3	820	14.5	99	1.7	179	3.2
Italy	IT	8614	91	1.1	2037	23.6	1084	12.6	463	5.4	74	0.9	4865	56.5
Lithuania	LT	1183	287	24.3	284	24	512	43.3	90	7.6	10	0.8	0	0
Luxembourg	LU	102	0	0	7	6.9	52	51	29	28.4	14	13.7	0	0
Latvia	LV	470	14	3	216	46	136	28.9	43	9.1	61	13	0	0
Malta	MT	9	4	44.4	1	11.1	2	22.2	2	22.2	0	0	0	0
Netherlands	NL	724	0	0	3	0.4	249	34.4	315	43.5	149	20.6	8	1.1
Poland	PL	5643	52	0.9	120	2.1	725	12.8	173	3.1	116	2.1	4457	79
Portugal	PT	1944	94	4.8	948	48.8	490	25.2	222	11.4	41	2.1	149	7.7
Romania	RO	3399	145	4.3	1875	55.2	1319	38.8	34	1	20	0.6	6	0.2
Sweden	SE	23418	2043	8.7	11065	47.2	8059	34.4	1617	6.9	506	2.2	128	0.5
Slovenia	SI	154	11	7.1	69	44.8	50	32.5	7	4.5	2	1.3	15	9.7
Slovakia	SK	1760	487	27.7	636	36.1	578	32.8	52	3	7	0.4	0	0
United Kingdom	UK	10961	441	4	3573	32.6	5216	47.6	1356	12.4	375	3.4	0	0

Table 1.2. Total number and percentage number of river water-bodies (including HMWBs) at high, good, moderate, poor, bad and unknown ecological status for each of the 28 EU member states. Data extracted from WISE (EEA, 2015).

Country	Country	Total Count	High	High (%)	Good	Good (%)	Moderate	Moderate (%)	Poor	Poor (%)	Bad	Bad (%)	Unknown	Unknown (%)
Austria	AT	7339	1311	17.9	1738	23.7	3806	51.9	389	5.3	78	1.1	17	0.2
Belgium	BE	455	7	1.5	115	25.3	102	22.4	89	19.6	112	24.6	30	6.6
Bulgaria	BG	688	31	4.5	270	39.2	227	33.0	93	13.5	67	9.7	0	0
Cyprus	CY	216	0	0	68	31.5	76	35.2	16	7.4	3	1.4	53	24.5
Czech Republic	CZ	1069	0	0	180	16.8	155	14.5	727	68.0	0	0	7	0.7
Germany	DE	9072	12	0.1	699	7.7	2644	29.1	3251	35.8	2211	24.4	255	2.8
Denmark	DK	16881	847	5.0	5923	35.1	5041	29.9	1295	7.7	536	3.2	3239	19.2
Estonia	EE	645	9	1.4	469	72.7	145	22.5	22	3.4	0	0	0	0
Greece	EL	1237	11	0.9	474	38.3	244	19.7	158	12.8	12	1.0	338	27.3
Spain	ES	94	10	10.6	13	13.8	7	7.4	13	13.8	4	4.3	47	50.0
Finland	FI	1602	139	8.7	390	24.3	303	18.9	86	5.4	31	1.9	653	40.8
France	FR	10824	732	6.8	3849	35.6	4423	40.9	1362	12.6	415	3.8	43	0.4
Croatia	HR	1231	262	21.3	364	29.6	270	21.9	183	14.9	152	12.3	0	0
Hungary	HU	869	0	0	68	7.8	295	33.9	184	21.2	37	4.3	285	32.8
Ireland	IE	4565	654	14.3	1823	39.9	1135	24.9	803	17.6	93	2.0	57	1.2
Italy	IT	7644	84	1.1	1925	25.2	1029	13.5	429	5.6	70	0.9	4107	53.7
Lithuania	LT	832	144	17.3	194	23.3	417	50.1	68	8.2	9	1.1	0	0
Luxembourg	LU	102	0	0	7	6.9	52	51.0	29	28.4	14	13.7	0	0
Latvia	LV	204	13	6.4	105	51.5	56	27.5	10	4.9	20	9.8	0	0
Malta	MT	0	0	0	0	0	0	0	0	0	0	0	0	0
Netherlands	NL	254	0	0	0	0	133	52.4	107	42.1	14	5.5	0	0
Poland	PL	4586	27	0.6	66	1.4	686	15.0	136	3.0	72	1.6	3599	78.5
Portugal	PT	1705	46	2.7	860	50.4	433	25.4	209	12.3	40	2.3	117	6.9
Romania	RO	3262	142	4.4	1857	56.9	1224	37.5	25	0.8	8	0.2	6	0.2
Sweden	SE	15563	1378	8.9	7176	46.1	5340	34.3	1202	7.7	379	2.4	88	0.6
Slovenia	SI	135	9	6.7	66	48.9	47	34.8	7	5.2	2	1.5	4	3.0
Slovakia	SK	1760	487	27.7	636	36.1	578	32.8	52	3.0	7	0.4	0	0
United Kingdom	UK	9080	201	2.2	2791	30.7	4508	49.6	1241	13.7	339	3.7	0	0

Table 1.3. Total length and percentage length of rivers (including HMWBs) at high, good, moderate, poor, bad and unknown ecological status for each of the 28 EU member states. Data extracted from WISE (EEA, 2015).

Country	Country	Total Length	High	High (%)	Good	Good (%)	Moderate	Moderate (%)	Poor	Poor (%)	Bad	Bad (%)	Unknown	Unknown (%)
Austria	AT	31393	4291	13.7	6970	22.2	17136	54.6	2341	7.5	606	1.9	47	0.1
Belgium	BE	9309	95	1.0	2435	26.2	2376	25.5	1756	18.9	2146	23.1	500	5.4
Bulgaria	BG	25569	862	3.4	8711	34.1	8962	35.1	4342	17.0	2691	10.5	0	0
Cyprus	CY	2579	0	0	842	32.6	1078	41.8	217	8.4	41	1.6	402	15.6
Czech Republic	CZ	21175	0	0	3409	16.1	3169	15.0	14111	66.6	41	0.2	446	2.1
Germany	DE	126158	152	0.1	9832	7.8	37117	29.4	41492	32.9	25022	19.8	12548	9.9
Denmark	DK	12047	926	7.7	4847	40.2	3851	32.0	821	6.8	336	2.8	1264	10.5
Estonia	EE	12107	295	2.4	8111	67.0	3218	26.6	481	4.0	0	0	0	0
Greece	EL	12719	133	1.0	4946	38.9	2689	21.1	1803	14.2	147	1.2	3002	23.6
Spain	ES	579	44	7.6	51	8.8	47	8.1	80	13.8	40	6.9	317	54.7
Finland	FI	28875	4659	16.1	7172	24.8	6146	21.3	2131	7.4	810	2.8	7960	27.6
France	FR	241684	10881	4.5	70305	29.1	114044	47.2	34454	14.3	10517	4.4	1480	0.6
Croatia	HR	13041	1800	13.8	4126	31.6	3597	27.6	2380	18.3	1138	8.7	0	0
Hungary	HU	18802	0	0	1907	10.1	8480	45.1	3704	19.7	709	3.8	4003	21.3
Ireland	IE	21037	1864	8.9	7514	35.7	6198	29.5	4472	21.3	597	2.8	391	1.9
Italy	IT	78812	655	0.8	16781	21.3	12497	15.9	6520	8.3	986	1.3	41373	52.5
Lithuania	LT	14251	2605	18.3	3585	25.2	6723	47.2	1255	8.8	83	0.6	0	0
Luxembourg	LU	2597*	0	0	191	7.4	1161	44.7	883	34.0	362	13.9	0	0
Latvia	LV	7752	535	6.9	3841	49.5	2113	27.3	426	5.5	835	10.8	0	0
Malta	MT	0	0	0	0	0	0	0	0	0	0	0	0	0
Netherlands	NL	4757	0	0	0	0	2336	49.1	2262	47.6	158	3.3	0	0
Poland	PL	111485	749	0.7	2352	2.1	24667	22.1	4076	3.7	2208	2.0	77431	69.5
Portugal	PT	598575	79628	13.3	136786	22.9	105583	17.6	192295	32.1	821	0.1	83463	13.9
Romania	RO	74473	2346	3.2	36766	49.4	33973	45.6	1185	1.6	177	0.2	26	0
Sweden	SE	79467	6181	7.8	33063	41.6	30044	37.8	7181	9.0	2346	3.0	651	0.8
Slovenia	SI	2619	168	6.4	1220	46.6	1039	39.7	117	4.5	26	1.0	50	1.9
Slovakia	SK	18944	3786	20.0	6384	33.7	7501	39.6	1141	6.0	133	0.7	0	0
United Kingdom	UK	99748	1653	1.7	28009	28.1	48209	48.3	17689	17.7	4190	4.2	0	0

Table 1.4. Total number and percentage number of lakes (including HMWBs) at high, good, moderate, poor, bad and unknown ecological status for each of the 28 EU member states. Data extracted from WISE (EEA, 2015).

Country	Country	Total Count	High	High (%)	Good	Good (%)	Moderate	Moderate (%)	Poor	Poor (%)	Bad	Bad (%)	Unknown	Unknown (%)
Austria	AT	62	21	33.9	38	61.3	3	4.8	0	0	0	0	0	0
Belgium	BE	18	0	0	0	0	9	50.0	4	22.2	3	16.7	2	11.1
Bulgaria	BG	43	5	11.6	16	37.2	9	20.9	6	14.0	6	14.0	1	2.3
Cyprus	CY	18	0	0	10	55.6	6	33.3	0	0	1	5.6	1	5.6
Czech Republic	CZ	71	0	0	13	18.3	0	0	54	76.1	0	0	4	5.6
Germany	DE	712	64	9.0	212	29.8	270	37.9	111	15.6	23	3.2	32	4.5
Denmark	DK	941	133	14.1	144	15.3	206	21.9	138	14.7	162	17.2	158	16.8
Estonia	EE	89	3	3.4	48	53.9	30	33.7	5	5.6	0	0	3	3.4
Greece	EL	50	0	0	1	2.0	11	22.0	7	14.0	3	6.0	28	56.0
Spain	ES	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a
Finland	FI	4275	540	12.6	745	17.4	387	9.1	80	1.9	18	0.4	2505	58.6
France	FR	439	2	0.5	70	15.9	82	18.7	53	12.1	40	9.1	192	43.7
Croatia	HR	34	7	20.6	5	14.7	3	8.8	14	41.2	5	14.7	0	0
Hungary	HU	213	5	2.3	32	15.0	25	11.7	9	4.2	5	2.3	137	64.3
Ireland	IE	807	314	38.9	209	25.9	255	31.6	17	2.1	6	0.7	6	0.7
Italy	IT	300	2	0.7	78	26.0	35	11.7	27	9.0	3	1.0	155	51.7
Lithuania	LT	345	143	41.4	90	26.1	90	26.1	21	6.1	1	0.3	0	0
Luxembourg	LU	0	0	0	0	0	0	0	0	0	0	0	0	0
Latvia	LV	259	1	0.4	111	42.9	77	29.7	29	11.2	41	15.8	0	0
Malta	MT	0	0	0	0	0	0	0	0	0	0	0	0	0
Netherlands	NL	450	0	0	3	0.7	103	22.9	206	45.8	135	30.0	3	0.7
Poland	PL	1038	25	2.4	54	5.2	37	3.6	30	2.9	34	3.3	858	82.7
Portugal	PT	122	1	0.8	62	50.8	48	39.3	7	5.7	0	0	4	3.3
Romania	RO	131	3	2.3	18	13.7	92	70.2	8	6.1	10	7.6	0	0
Sweden	SE	7232	653	9.0	3794	52.5	2278	31.5	354	4.9	117	1.6	36	0.5
Slovenia	SI	13	1	7.7	1	7.7	3	23.1	0	0	0	0	8	61.5
Slovakia	SK	0	0	0	0	0	0	0	0	0	0	0	0	0
United Kingdom	UK	1119	62	5.540661	424	37.9	489	43.7	111	9.9	33	2.9	0	0

Table 1.5. Total area and percentage area of lakes at high, good, moderate, poor, bad and unknown ecological status for each of the 28 EU member states. Data extracted from WISE (EEA, 2015).

Country	Total Area	High	High (%)	Good	Good (%)	Moderate	Moderate (%)	Poor	Poor (%)	Bad	Bad (%)	Unknown	Unknown (%)
AT	934	91	9.7	839	89.8	3	0.3	0	0	0	0	0	0
BE	41	0	0	0	0	12	29.3	6	14.6	21	51.2	2	4.9
BG	76	2	2.6	41	53.9	6	7.9	10	13.2	13	17.1	2	2.6
CY	28	0	0	8	28.6	17	60.7	0	0	0	0	3	10.7
CZ	249	0	0	37	14.9	0	0	207	83.1	0	0	5	2.0
DE	2400	255	10.6	1087	45.3	725	30.2	262	10.9	35	1.5	33	1.4
DK	467	53	11.3	51	10.9	132	28.3	69	14.8	129	27.6	31	6.6
EE	1965	6	0.3	348	17.7	1538	78.3	70	3.6	0	0	4	0.2
EL	1051	0	0	20	1.9	569	54.1	236	22.5	59	5.6	165	15.7
ES		n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a	n/a
FI	28172	7225	25.6	14499	51.5	2687	9.5	321	1.1	46	0.2	3395	12.1
FR	1965	1	0.1	379	19.3	816	41.5	446	22.7	58	3.0	264	13.4
HR	169	10	5.9	35	20.7	8	4.7	90	53.3	26	15.4	0	0
HU	1267	5	0.4	824	65.0	125	9.9	23	1.8	15	1.2	274	21.6
IE	2629	983	37.4	628	23.9	979	37.2	18	0.7	16	0.6	4	0.2
IT	2158	0	0	940	43.6	409	19.0	302	14.0	14	0.6	492	22.8
LT	809	364	45.0	180	22.2	187	23.1	66	8.2	12	1.5	0	0
LU	0	0	0	0	0	0	0	0	0	0	0	0	0
LV	825	1	0.1	344	41.7	168	20.4	121	14.7	189	22.9	0	0
MT	0	0	0	0	0	0	0	0	0	0	0	0	0
NL	3046	0	0	3	0.1	2217	72.8	522	17.1	303	9.9	1	0
PL	2294	47	2.0	114	5.0	103	4.5	129	5.6	112	4.9	1787	77.9
PT	741	0	0	298	40.2	421	56.8	3	0.4	0	0	21	2.8
RO	993	0	0	412	41.5	518	52.2	12	1.2	50	5.0	0	0
SE	29191	2062	7.1	13944	47.8	10900	37.3	1787	6.1	249	0.9	249	0.9
SI	37	3	8.1	20	54.1	8	21.6	0	0	0	0	6	16.2
SK	0	0	0	0	0	0	0	0	0	0	0	0	0
UK	1934	145	7.5	573	29.6	546	28.2	204	10.5	468	24.2	0	0

Table 1.6. Total numbers of all surface water-bodies (rivers, lakes, coastal and transitional, including HMWBs) at good, fail, or unknown status for chemistry for each of the 28 EU member states. Data extracted from WISE (EEA, 2015).

Country	Country	Total Count	Good	Good (%)	Fail	Fail (%)	Unknown	Unknown(%)
Austria	AT	7401	7361	99.5	18	0.2	22	0.3
Belgium	BE	560	194	34.6	168	30	198	35.4
Bulgaria	BG	759	596	78.5	23	3	140	18.4
Cyprus	CY	260	193	74.2	12	4.6	55	21.2
Czech Republic	CZ	1140	803	70.4	330	28.9	7	0.6
Germany	DE	9863	8695	88.2	811	8.2	357	3.6
Denmark	DK	17984	32	0.2	55	0.3	17897	99.5
Estonia	EE	750	746	99.5	4	0.5	0	0
Greece	EL	1575	347	22	147	9.3	1081	68.6
Spain	ES	175	0	0	0	0	175	100
Finland	FI	6153	3938	64	27	0.4	2188	35.6
France	FR	11523	4965	43.1	2627	22.8	3931	34.1
Croatia	HR	1315	1279	97.3	34	2.6	2	0.2
Hungary	HU	1082	35	3.2	28	2.6	1019	94.2
Ireland	IE	5670	1603	28.3	41	0.7	4026	71
Italy	IT	8614	1521	17.7	411	4.8	6682	77.6
Lithuania	LT	1183	1169	98.8	14	1.2	0	0
Luxembourg	LU	102	71	69.6	31	30.4	0	0
Latvia	LV	470	29	6.2	0	0	441	93.8
Malta	MT	9	0	0	0	0	9	100
Netherlands	NL	724	506	69.9	179	24.7	39	5.4
Poland	PL	5643	152	2.7	279	4.9	5212	92.4
Portugal	PT	1944	798	41	11	0.6	1128	58
Romania	RO	3399	3165	93.1	228	6.7	6	0.2
Sweden	SE	23418	3	0	23415	100	0	0
Slovenia	SI	154	146	94.8	7	4.5	1	0.6
Slovakia	SK	1760	1673	95.1	87	4.9	0	0
United Kingdom	UK	10961	3910	35.7	181	1.7	6870	62.7

Interpretation of data within the WISE-WFD database is very much dependant on which parameters are used. For example, Slovenia has 6.7 % of its rivers designated as high ecological status, but this relates to only nine 9 river sites, whereas France has 6.8% of its rivers designated as high which corresponds to 732 rivers (Table 1.2). Additionally, if measurement is used instead of numbers of, a different set of results are generated. If length is used as the determining factor then 4.5 % of rivers in France are at high ecological status, corresponding to 10881 km, whereas Slovenia has 6.4 % of its rivers at high ecological status corresponding to 168 km (Table 1.3). Similarly, Austria has 33.9 % of its lakes designated as high ecological status, but this relates to only 21 lake sites, whereas Finland has 12.6% of its sites designated as high ecological status which corresponds to 540 lake sites (Table 1.4). Again, if a measurement parameter is incorporated into the lake analysis (area for lakes) proportions again vary (Table 1.5). Therefore interpretation of data is based on one of four concepts: 1) the total number of sites at high status; 2) the percentage of sites at high status based on the number of sites at high status relative to the total number of sites; 3) the total measurement of sites at high status (length of rivers, area of lakes); or 4) the percentage of sites at high status based on the total measurement of sites at high status relative to the total overall measurement.

Additionally, some sources, for example the WISE-WFD database, include both natural and heavily modified water-bodies (HMWBs) in the total number of surface water-bodies, whereas others, for example individual member states, exclude HMWBs, thereby making percentage comparisons difficult. Interpretation is important because it may influence how resources are directed towards management, as well as determining how data are presented to other MS. White et al. (2014), for example, misinterprets percentage of sites as number of sites, and reports France and the UK to have low to no high status sites, while Greece is reported to have the highest number of sites. In reality, both France and the UK have 550 and 250 more sites designated as high status than Greece, respectively.

Based on numbers, Sweden (2043), Austria (1332), Ireland (1012), Denmark (980), France (747) and Finland (681) all have over 500 sites designated as high ecological status (Table 1.1). However, almost 100% of sites in Sweden fail based on chemical status, while for many of the other countries the chemical status is only partially known (Table 1.6).

1.4. Management strategies for high status water-bodies – case studies

1.4.1. Sweden

Sweden has five main River Basin Districts (RBDs) (SE1 – SE5), and five smaller international RBDs that are managed within the River Basin Management Plans (RBMPs) of the larger RBDs (European Commission, 2012c). Two of these RBDs (SE1 and SE2) and the SE1 incorporated SE1TO are responsible for 95% of Sweden's high status sites, although both RBDs are considerably larger than either of the other three RBDs. The national priority in Sweden is for water-bodies to achieve "good" status, while actions to maintain high status are dealt with at the local scale, in the municipalities in which they occur (Fredrick Gummarsson, Water Unit Sweden, personal communication). However, data for 2015 (VISS, 2015) show that the number of high status sites in Sweden has increased from 2037 in the last cycle (2009 – although this figure differs from the 2043 figure presented by the WISE-WFD database, 2012) to 2420 in the most recent cycle (2015). This equates to an overall increase of 383 sites, (i.e. there may be both increases and decreases combined within this figure). An overall increase of 391 sites were reported within the SE2 or Bothnian Sea Bay RBD, which at the very least implies deteriorations of eight sites within the other RBDs. The main reasons for these increases (and possibly declines) have been attributed to “more knowledge” and changes in approaches to classifying water-bodies, either through amended guidelines especially for hydro-morphological assessment, or through increased use of expert judgement, where, for example, data were either not available or considered unreliable (Fredrick Gummarsson and Bart De Wachter, Water Unit Sweden, personal communication). The European Commission (2012c) lists diffuse pollution from agriculture and forestry, stand-alone housing, and atmospheric deposition, as affecting up to 100% of water-bodies in the SE1 and SE2 RBDs, while flow and morphological modifications, as well as river management issues are also significant pressures. Mercury is the main hazardous substance responsible for the 100% of chemical fails (European Commission, 2012c; VISS, 2015). In contrast, Österberg (no date), lists habitat modification as a result of changes to flow, connectivity and morphology as being the biggest environmental challenge in the SE1 - Bothnian Bay Water District, with eutrophication impacting only 19% of coastal waters and 5% and 1% of rivers and lakes respectively.

The RBD SE1 (+ SE1TO), or Bothnian Bay Water District, contains 1494 High ecological status sites which equates to *ca*. 22% of the water-bodies in this district (data extracted from WISE-WFD database). The majority of these high status sites are located in the North of the district, where the pressures mainly relate to forestry and hydro-power production (European Commission, 2012c). While proposals for a management plan for the Bothnian Bay Water District have been drawn up (Österberg, no date), this does not specifically set out targets to maintain high status sites. However, the report states that although up to 45% of the water-bodies in SE1 are at risk of failing to meet good ecological status or at risk of deteriorating below their current standing, these risks are only applicable to sites not already at good status (Österberg, no date). As the proposed "Programs of Measures" within the report, are generally aimed at improving water-bodies at below good status, it is unclear if they are likely to have any knock-on benefits for sites already at good or high status.

1.4.2. Austria

In Austria, ca. 98% of the high status sites are within the Danube RBD (AT1000) (data extracted from WISE-WFD database). At over 80,000 km^2 this RBD area is far in excess of the other two Austrian RBDs. Here, water abstractions and diffuse pollution from agriculture, forestry, de-icing materials used in airports, mining and contaminated sites are the main pressures impacting 56% and 16% of water-bodies respectively, while 38% of water-bodies are under no pressures (European Commission, 2012d). Of the water-bodies within AT1000, 644 are protected either for drinking water abstraction (210) (under Article 7 of the WFD), as bathing waters (251), or for their bird (50), fish (67) and habitat (86) compositions, although this is made up of both surface and ground waters (European Commission, 2012d). Unlike Sweden, 99.5% of the water-bodies in Austria are at good chemical status (Table 1.6). A comparison of the percentage number of high status rivers and lakes reported in the Nationaler Gewässerbewirtschaftungsplan (NGP) 2009 (National Water Management Plan 2009) (NGP, 2010) against data reported in the draft Nationaler Gewässerbewirtschaftungsplan 2015 (NGP, 2015), (Tables 1.7 and 1.8, respectively), shows that, while there has been a slight improvement in the percentage number of high status river sites, the number of high status lake sites, especially in the Danube RBD, has declined by 16%. These declines, as with Sweden, have been attributed to incorporating additional techniques into the survey method (for example, the assessment of hydro-geomorphology) that were not part of the original survey methods carried out in 2009 (NGP, 2015; Stephan Nemetz, Umweltbundesamt, Austria, personal communication). Section 6.2 of the NGP (2010) lists some of the measures applicable in Austria in order to prevent deteriorations of HSWs. It stated that, while some deteriorations to high status sites may result from point or diffuse sources of pollution, or from hydro-morphological changes to water-bodies as a result of flood protection schemes or the development of hydropower plants, restrictions are in place to limit these pressures, and they are only allowable if their absence is likely to cause significant impairment to the public (NGP, 2010).

Table 1.7. Percentage of river sites in each river basin district in Austria at high status in 2009 and 2015. Data from: NGP (2010) Page 70 & NGP (2015) page 124.

Rivers	High Status 2009 (%)	High status 2015 (%)
Austria (overall)	14	15
Donau	14	15
Rhein	11	15
Elbe	3	5

Table 1.8. Percentage of lake sites in each river basin district in Austria at high status in 2009 and 2015. Data from: NGP (2010) Page 72 & NGP (2015) page 125.

Lakes	High Status 2009 (%)	High status 2015 (%)
Austria (overall)	34	16
Donau	34	18
Rhein	-	
Elbe	No natural lakes	

1.4.3. Ireland

Ireland had a total of seven RBDs[3] of which three were international being shared with Northern Ireland. Three of the RBDs, IEWE, IESW, and GBNIIENW accounted for 88.4% of the high status water-bodies, while the RBD GBNIIENB is the only RBD to have had zero high status sites (data extracted from WISE-WFD database; EEA, 2015). Point source pollution from waste water treatment plants and diffuse pollution from agriculture and faulty septic tanks were listed as the largest pressures, while greater than 40 % of water-bodies in each RBD IEWE, IESW, and GBNIIENW experienced no significant pressures (European Commission, 2012e). Many surface and ground water-bodies occur in areas protected for: drinking water abstractions (943); birds (136); fish (31); habitats (426); and bathing waters (126); amongst others (European Commission 2012e). However, an Irish Environmental Protection Agency "key indicators of the aquatic environment" report (EPA, 2009) noted large declines in the number of high status river sites between the years 1987-2008, and, although information for lakes and transitional waters has been listed as uncertain, they are likely to be following the same trend (Ní Chatháin et al., 2012). Indeed, a more recent extraction of data from the EPA Geo-portal website (2015), reveals that of 1822 river sites recorded as high in 2009, only 827 sites were recorded as high in the subsequent sampling period 2010-2012, with deteriorations to good (289), moderate (109) and poor (24) occurring in 422 sites, while 570 sites had their status unassigned. On the other hand, 950 river sites which were recorded as good (482), moderate (83), poor (18), bad (1), pass (30), or not monitored (336) in 2009, improved to high for the 2010-2012 period. Of 102 "monitored" lake sites recorded as high in 2009, only 23 were

[3] For the second cycle of the River Basin Management Plan (2018-2021) a single national River Basin District (RBD) has been defined (RBMP,2018). This single RBD has being further divided into 46 catchment management units, with these units being divided again further into 583 sub-catchments (RBMP, 2018). (See also Chapter 5 for more details regarding the 2018-2021 RBMP).

recorded as high in the 2010-2012 sampling period, with deteriorations to good (27) and moderate (3) occurring in 30 sites, while 49 sites had either no corresponding data or were unassigned (data extracted from EPA Geo-portal website, 2015). Additionally, 26 monitored lake sites, which were recorded as good (19) or moderate (7) in 2009, improved to high for the 2010-2012 period (data extracted from EPA Geo-portal website, 2015).

Within the RBMPs, preventing deteriorations was highlighted as a core objective, along with achieving the objectives of protected areas, especially in relation to water quality and the protection of species such as the Freshwater Pearl Mussel (*Margaritifera margaritifera*) (WRBMP, 2010; NWIRBMP, 2010; SWRBMP, 2010). The POMs within these RBMPs included targeting: point source pollution from urban waste water discharges, and un-sewered connections through appropriate treatment systems and septic tanks (WRBMP, 2010; NWIRBMP, 2010; SWRBMP, 2010). Agricultural pollution was targeted through the Good Agricultural Practice Regulations (updated to SI 65 of 2018) and Nitrate Regulations, while measures aimed at targeting Natura 2000 sites, especially with regard to *M. margaritifera* and shellfish waters, were to be implemented by local authorises and through Pollution Reduction Programmes (PRP), respectively (WRBMP, 2010; NWIRBMP, 2010; SWRBMP, 2010). Other measures targeted pollution from forestry, pesticide use, pressures from aquaculture and peat extraction, and the sale of invasive non-native species (INNS), while measures regarding flood management and potential future threats under climate change were also mentioned (WRBMP, 2010; NWIRBMP, 2010; SWRBMP, 2010). Furthermore, each water body had its own Water Management Unit Action Plan (available at www.wfdireland.ie) that further delineated suitable measures.

The scale of the observed declines in Ireland, however, prompted an additional "Management Strategies for the Protection of High Status Water Bodies" report (Ní Chatháin et al., 2012) to be drawn up as a benchmark statement. This report attributed the declines to stressors such as point source pollution or unintentional discharges, while low intensity practices such as: land–use change through drainage or fertilizer addition; one-off housing with poorly functioning septic tanks; deforestation and afforestation practices and associated drainage; and construction works and wind farm developments; are also cited as important factors, along with livestock accessing water-bodies and pollution from sheep dip. The report goes on to suggest key management strategies that include: defining the borders of high status catchments though the use of GIS, which should then be incorporated into planning and decision making processes carried out by all local and public authorities; the setting up a spatial network of high status sites akin to that of the Habitats Directive (OJEC, 1992) for protected habitats, and the restoration of previous high status sites, especially in RBDs that have experienced large declines, and in areas where few high status sites remain; and adopting additional measures from other EU Directives such as the Birds (OJEC, 1979; 2009) and Habitats Directives, the Nitrates Directive (OJEC, 1991a), the Urban Waste Water Treatment Directive (OJEC, 1991b), and the Environmental Impact Assessment Directive (OJEC, 1985) amongst others. Carrying out catchment walk-overs to evaluate potential pressures; screening planning applications for proposed developments within high status catchments; mapping drainage channels within high status catchments; education programmes; and increasing sampling frequency, especially for nutrients are also suggested. These management strategies are further discussed by White et al. (2014). It is suggested, that if fully implemented, these measures should abate the observed declines, although a coordinated effort from both local and public authorities, as well as local level and site-specific actions are required (Ní Chatháin et al., 2012). Additionally, through the detailing of two case studies, the report highlights the benefits of additional protection measures afforded under the Habitats Directive, especially in relation to deterring unregulated activities such as peat cutting, or construction works.

1.4.4. UK - Scotland

In the UK, 96% of the high status sites occur in the Scotland RBD UK01 (data extracted from WISE-WFD database), which comes under the stewardship of the Scottish Environmental Protection Agency (SEPA). Based on 2013 figures Scotland has 10.7% of its total water-bodies (surface and coastal) at high status (data extracted from Environment Scotland, 2015). However, of the 180 river sites designated as high in 2010, only 82 were high in 2013, with deteriorations to good (81), moderate (15) and poor (2) occurring in the other 98 sites (data extracted from Environment Scotland, 2015). On the other hand 69 river sites which were at good (54), moderate (10), poor (4) or bad (1) in 2010 improved to high in 2013. Similarly, of the 52 lakes recorded as high in 2010, only 32 were high in 2013, with the other sites deteriorating to good (21) and moderate (1), while 27 lake sites which were at good (24), moderate (1) or poor (2) in 2010 improved to high in 2013. For seven river sites fluctuations in status, i.e. high to good to high, were recorded in the intervening years 2011 and 2012. Agricultural diffuse pollution, physical modification of the water-bodies especially as a result of land use practices and dams, toxic substances from road run-off and urban diffuse pressures, and contaminated land are listed as significant water-body pressures (Natural Scotland, 2014). Also, it should be noted that some heavily modified water-bodies (HMWBs) are classified as high in Scotland, with at least two recorded as high by the European Commission (2012f).

Although not specific to HSWs, the 2009 RBMP for Scotland (SEPA, 2009) provides some information with regard to preventing deteriorations. These include measures aimed at assessing potential changes in agricultural land use practices in response to climate change and restricting land use and land cover change proposals likely to cause environmental effects; reducing sources of diffuse pollution through the possible use of legislative, economic (targeting farmer payments) and "education and advice" actions; and the assessment of developments by SEPA, Scottish Water and local Authorities based on the holding capacity of local sewage and treatment works, and the potential chances of deteriorations to occur due to increased or cumulative discharge levels (RBMP Scotland, 2009). Measures targeting mining and quarrying activities; the use of the sustainable urban drainage system (SUDS) to address urban diffuse pollution; the relocation of fish-farm and aquaculture activities away from sensitive areas; and better management of forestry activates (RBMP Scotland, 2009) are also proposed. Reducing deteriorations from flow and water abstractions include targeting hydropower operators, managing public drinking water demands, and managing agricultural irrigation levels (RBMP Scotland, 2009). Other measures include the management of riparian vegetation through the use of buffer zones and removal of invasive species; the management of engineering stressors; ensuring adequate fish passes are present; and controlling the threats from INNS (RBMP Scotland, 2009). As with Ireland, each water body has its own water body data sheets (available at http://www.sepa.org.uk/environment/water/river-basin-management-planning/publications#Measures) that further delineates suitable measures.

Additional proposals aimed at preventing deteriorations in Scotland are presented within the "current condition and challenges for the future: Scotland river basin district" report (Natural Scotland, 2014). This report further details efforts to target diffuse pollution, including: engaging with land managers to "identify and reduce" risks; re-assessing the targeting of funds to better control pollution sources, for example, by creating buffer zones; the provision of integrated advice to land managers as well as training in relation to "good farming practice"; and assessing the potential to reduce P levels in livestock feed. Water-bodies that are close to the bottom level of classification have been targeted, as well as those at risk from invasive species (Natural Scotland, 2014). Action plans specific to rural diffuse pollution (DPMAG, 2012) and INNS (Natural Scotland, 2013) are also useful.

1.5. Discussion and conclusions

Adoption of the EU WFD by MS has been a considerable challenge to date, requiring time, resources and funding (Hering et al., 2010). While the main focus so far has been for countries to achieve good ecological status, the protection of HSWs has not being regarded as a key issue (White et al., 2014). As water quality improves, the next logical step is progression towards the high status goal. In many instances this may be difficult (e.g. 79% of German water-bodies are subject to a target date extension; European Commission, 2012g), or even if restoration is possible, the time and financial costs involved may be considerable and the end results are likely to fall short of expected targets (Benayas et al., 2009; Bullock et al. 2011). Countries that already have water-bodies at high status should, therefore, prioritise the no deterioration objective which in the longer term is likely to prove more cost effective, e.g. targeting the small impacts that lead to high status deteriorations versus large scale restoration efforts (White et al., 2014; Irvine et al., 2011). Additionally, Doody et al. (2014) suggest that prioritising some water-bodies, as opposed to all water-bodies, for protection, may be the "least worst" option in the face of threats to aquatic ecosystems arising from the likelihood of increased global food demands and associated agricultural intensification. In light of their benefits, prioritising the protection of HSWs is perhaps the best approach.

While Ireland and Scotland have attempted to address the no deterioration objective through POMs and additional action plans, there are still large numbers of deteriorations occurring. Nevertheless, large numbers of improvements from good or below to high are also occurring, with many fluctuations in between according to their respective monitoring protocols. Indeed, in Sweden and Austria, improvements and declines have been attributed to changes in monitoring techniques, either through adjusted guidelines or through the incorporation of previously unused sampling methods. Changes in classification due to improved knowledge are also likely to be replicated in many other MS. In Latvia for example, one of the two high status sites reported for 2009 has deteriorated, with this deterioration being attributed to the employment of a fuller suite of biological quality elements as called for by the WFD, as opposed to the incomplete suite used in 2009, when the first river basin management plans were approved (Rudīte Vesere, Director, Environmental Protection Department, Latvia, personal communication).

In Ireland and Scotland, declines have mainly been attributed to agricultural pressures, while the reasons for improvements may be due to the implementation of the POMs and management strategies. However, Natural Scotland (2014) suggests that, as of the end of 2012, there was only likely to be a small improvement in water quality conditions resulting from the implementation of POMs, due in part to a lag time between POMs being employed and any observed response from aquatic communities. The frequency of sampling may be a factor in the large numbers of deteriorations and improvements, as Scotland monitors water-bodies every year and then provides an average result over a number of years (Environment Scotland, 2015; WISER, 2016), whereas Ireland, using the Q-value system, sample a water body once every three years (EPA, 2006). Additionally, monitoring methods that are less quantitative and more subjective, such as Ireland Q-value system, may be more prone to operator bias. This compares to methods, for example, employed by SEPA and the Environment Agency (UK), that routinely involve lab identification and quantification of taxa collected and the generation/calculation of biotic indices such as the Biological Monitoring Working Party (BMWP) index. As with other MS, changes in monitoring techniques, especially with regard to assessing hydro-morphology may also play a role (Natural Scotland, 2014), while the possible influence of climate change or natural climate phenomena such as the North Atlantic Oscillation (NAO) should not be ruled out (Wilby et al., 2006; Jennings et al., 2009; Mellander et al., 2018).

Although, the WFD was first introduced in 2000, and 2015 was initially set as the target year to achieve the WFD aims and objectives, there are still some issues that need to be resolved (Reyjol et al., 2014). While the inter-calibration process aims to allow comparisons between MS (Heiskanen et al., 2004; Bennett et al., 2011), there is still not a holistic approach to implementing and assessing the WFD objectives across MS (Birk et al., 2012; Pardo et al., 2012) or, as highlighted here, a standard method for presenting data. Additionally, within MS it is apparent that the required suite of monitoring techniques are still in the process of trial and development (Cuadrado et al., 2014; Lewin et al., 2014), or in the case of climate change (Wilby et al., 2006), and lakes, transitional waters and the assessment of hydrological impacts (Reyjol et al., 2014), yet to be defined. River Basin Management Plans from 2009 should have established a baseline year, from which any improvements or deteriorations were to be assessed against. However, this has been offset, as observed here, due to adjustments to monitoring techniques and late implementation of required sampling methods. This is especially true given the late adoption of RBMPs by several MS (European Commission, 2012a) and, therefore, the probability of an even longer lag time before any improvements are to be seen, while in some instances the approaches of the 2009 RBMPs, may have severely under-estimated the scale of pressures (Natural Scotland, 2014).

The management strategies for HSWs put forward by Sweden and Austria, appear small scale or non-existent in comparison to those adopted by Ireland and Scotland. While there are many similarities between the Irish and Scottish methods for targeting deteriorations, this is hardly surprising given their similarities in terms of economies, population sizes, and agricultural practices. Where differences do exist - such as the proposal to target P levels in animal feed, and appraising the potential switch from grassland to arable land as a result of climate change in Scotland; or the delineation of HSW catchments through GIS which should then be used by local authorises in any planning assessments, or adopting additional approaches and measures from other EU directives in sites otherwise outside the realm of protected status, in Ireland - each country may learn from the other. Switches to arable land are likely to increase pressures in Ireland also, for example, although to date this has received little attention (Doody et al., 2012). Across Europe it is likely that many HSWs and/or reference sites are located in protected areas (e.g. Mayes and Codling, 2009; Cuadrado et al., 2014; Lewin et al. 2014), and therefore some guidelines already developed for water-bodies in protected sites may be transferable to other HSWs.

Consideration should also be given to other solutions. Doody et al. (2012) for example, proposes the use of critical source areas (CSAs) as a cost effective method to target diffuse sources of P pollution in Ireland, especially in relation to HSWs. These CSAs are relatively minor sections within a field, farm or catchment that are responsible for the majority of pollution transfers to water-bodies (McDowell et al., 2014; Thomas et al., 2016). By targeting CSAs, therefore, it should be possible to reduce the majority of discharges with an optimum of effort. Implementing the process involves modelling the catchment to identify CSAs and then managing pressures from these CSA areas through the use of, for example, riparian buffer zones. McDowell et al. (2014) for example, demonstrated the benefit of targeting CSAs as a cost-effective methods for reducing N and P losses in French and New Zealand catchments, respectively. Additionally, the most recent incarnation of the agri-environmental scheme in Ireland, Green Low-carbon Agri-Environmental Scheme (GLAS), which sets objectives aimed at reducing pressures on biodiversity and water quality, and from climate change, is specifically targeted at farms located in high status catchments (DAFM, 2015a; DAFM, 2015b), and, although voluntary, agri-environment schemes such as this should be encouraged where suitable. Whitehead et al. (2009) reviewed the potential consequences of climate change on the aquatic environment, assessing impacts to flow regimes, nutrient discharge levels and the probability of longer growing seasons, while Wilby et al. (2006) provides a list of key areas that require further research. Although both the Irish and Scottish and RBMPs to some degree address the issue of climate change, POMs should be continuously updated to reflect any new research (e.g. Jiménez et al., 2018). This is also the case for INNS.

In conclusion, HSWs are sensitive areas that require special attention. Here some of the issues they face are presented. While some countries have developed strategies for their protection, it is as yet unclear how effective these strategies have been or are likely to be. On the other hand, other countries appear more concerned with achieving the good status objective, which in the long run may be counter-productive. This review has highlighted the importance of HSWs in the EU with a view towards better understanding of the reasons for their declines (or improvements) and the requirement for more effective management strategies for their protection. With this in mind, there is a clear requirement, amongst other things, to provide more detailed risk assessments of HSWs that account for subtle and acute pressures and also account for more detailed monitoring in space and time.

In light of the recently observed declines in the number of HSWs in Ireland, and with the potential for these declines to be associated with changes in land use activities and land cover types, along with changes to hydrological regimes, and increased sediment pressures (Ní Chatháin et al., 2012; White et al., 2014), the subsequent chapters set out to investigate the reasons for these declines. To this end, Chapters 2, 3 and 4 assess the potential for HSW deteriorations to be caused by land use and land cover change, hydrological (streamflow) modifications, and sediment pressures, respectively. Following on from this, Chapter 5 presents a synopsis of the three previous chapters, as well as providing key recommendations to prevent future deteriorations.

Chapter 2.

2. The relationship between land cover change and high status water-bodies

2.1. Introduction

Freshwater ecosystems, such as rivers and lakes, are important because they support a disproportionally high level of biodiversity (Dudgeon et al., 2006; Strayer and Dudgeon, 2010; Dijkstra et al., 2014), as well as providing ecosystem services, such as clean drinking water, recreational facilities and associated economic incomes (Costanza et al., 1997; White et al., 2014). However, they are perhaps one of the world's most endangered ecosystems, being threatened by pollution stressors, flow alterations, habitat loss, invasive species, unsustainable use and climate change (Nilsson et al., 2005; Dudgeon et al., 2006; Poff and Zimmerman, 2010; Vörösmarty et al., 2010; Collen et al., 2014). High status water-bodies (rivers and lakes, and transitional and coastal waters - HSWs), as designated under the European Union (EU) Water Framework Directive (WFD) (OJEC, 2000), are near-natural water-bodies, that represent conditions largely un-impacted by anthropogenic activities (WG 2.3, 2003; Mayes and Codling, 2009). As high status water-bodies contribute significantly to the overall species diversity of catchments (Hering et al., 2010; White et al., 2014), they should be especially highly regarded and protected (Doody et al., 2012).

Research has documented the role that land use and land cover surrounding a water-body plays in determining its quality status (Wang, 2001; Jordan et al., 2012; Poole et al., 2013; Lange et al., 2014a; 2014b). Allan (2004), for example, describes how changes to natural geomorphic processes, such as modifications to erosion and deposition cycles by anthropogenic actions at the landscape level, are likely to impact on the dynamics and health of a river, to the detriment of the rivers' abiotic and biotic environment. Additionally, Wang (2001) discusses the importance of the catchment (i.e. land draining into a water body) when implementing water policies, and studies at the catchment scale have steadily increased in importance (Allan et al., 1997; Sliva and Dudley Williams, 2001; Lange et al., 2014a). Land use and land cover, especially related to agriculture (Foley et al., 2005; Jordan et al., 2005a; Moss, 2008; O'Dwyer et al., 2013; Smith et al., 2013; Glendell and Brazier, 2014) and forestry (Ahtiainen and Huttunen, 1999; O'Driscoll et al., 2011), are particularly associated with water quality degradation, although the potential impact from urbanisation (Miserendino et al., 2011; Shepherd et al., 2011) and from for example, peri-urban sewage treatment works and rural septic tank systems (Withers et al., 2014), is also considerable.

Agricultural systems, in contrast to undisturbed natural ecosystems, are leaky systems, transferring phosphorus (P) and nitrogen (N) from land to water (Hooda et al., 2000; Schröder et al., 2004). This may alter ecosystem functioning, and require ongoing intensive mitigation and management (Moss, 2008). For example, diffuse nutrient transfer from agricultural sources results in eutrophication of associated water-bodies, and is a persistent problem across Europe (EEA, 2005; EEA, 2012; van Dijk et al., 2016) and elsewhere (e.g. China - Le et al., 2010; United States - Dodds et al., 2009; Bhaduri et al., 2000; New Zealand - Monaghan et al., 2007; Matthaei et al., 2010). The diffuse nature of nutrient transport and the connection to storm events make any mitigation difficult (Withers et al., 2014). However, more recent data in the EU (1992-2012) have reported declines for P and N levels entering rivers, and P in lakes, with these being attributed to a reduction in agricultural inputs, the removal of P from detergents and improvements in the management of wastewater (EEA, 2015).

Nutrient loadings are only one pressure however, with land use and land cover changes also being related to changes to hydrology and subsequent flow patterns (Bhaduri et al., 2000; Malmqvist and Rundle, 2002) and sedimentation through soil erosion (Allan et al., 1997; Jordan et al., 2005b, Kasai et al., 2005; Scheurer et al., 2008). Hydrology may be influenced by agriculture through reduced vegetation cover, which has impacts for the soil surface layer and subsequent erosion (Sutherland et al., 2010), while drainage networks may increase run-off and result in higher flood flows and suspended sediment exports (Jones and Holmes, 1985; Blann et al., 2009). Similarly, forestry and urbanisation are associated with altering the hydrological regime, with physical impairment of habitat occurring through road crossings, drainage, hard surfaces and channelization (Jones and Holmes, 1985; Löfgren et al., 2009).

Sediment exports from agricultural land use practices are primarily associated with arable farming methods, such as row-cropping, along with over-grazing and poaching of bank-side areas by livestock (Waters, 1995; Evans et al., 2006). Despite the promotion of measures to reduce these threats, such as the use of contour ploughing and preventing cattle accessing waterways (e.g. agri-environmental schemes), sedimentation continues to result in aquatic degradations (Matthaei et al., 2006; Sutherland et al., 2010; Bilotta et al., 2012; Sutherland et al., 2012; Glendell et al., 2014; Ramezani et al., 2014). Forestry, mining, and urban development are other land use actives that serve as major sources of sedimentation (Waters, 1995). Additionally, land use practices may impact on aquatic ecosystems by acting as sources of pathogens

(Sliva and Dudley Williams, 2001; Buck et al., 2004; Monaghan et al., 2007), pesticides, heavy metals and invasive species (Jones and Holmes, 1985; Malmqvist and Rundle, 2002).

The HSW designation is pertinent to all EU member states, although Ireland, on the Atlantic western fringe of Europe, has a particularly rich distribution of HSWs. However, significant declines in the numbers of high status river sites have been noted; with lakes and transitional waters likely following a similar trend (White et al., 2014). Reasons for these declines have been attributed to changes in land use and land cover trends, especially associated with agricultural practices, while additional environmental concerns such as climate change and related extreme flooding events are also potential factors (Ní Chatháin et al., 2012).

With this background, and with regard to the decreasing condition of HSWs in Ireland as a case study, the aim of this study was to investigate the relationship between changes to land use and land cover, and the high status classification of adjacent water-bodies. The objectives were to: 1) use readily available spatial datasets to compare changing land cover trends with trends in HSWs; and 2) investigate the relationship between declines in high status and land cover tested under the null-hypothesis that: there was no relationship between declines in high status and noted changes in adjacent land use and land cover.

2.2. Methods

2.2.1. Study Sites

Ireland assesses river water quality for the WFD objectives, primarily through the use of aquatic invertebrates, whereby a “Q-value” score of between 1 (bad water quality) to 5 (high water quality), which is related to the sensitivity of invertebrates to stressors, is assigned (EPA, 2007). Factors impacting on these invertebrates are therefore likely to result in a decline in water-body status. The sites used in this study were high status river sites in Ireland that, based on Environmental Protection Agency (EPA) ecological quality (Q-value) monitoring, that had either Lost their high status (e.g. gone from high to good, moderate, poor or bad); had consistently Maintained their high status; or had Gained in status (e.g. from good to high). An original dataset of 654 high status sites were edited to exclude sites that:

1. Occurred on streams of the order of one and greater than four (as the mean stream order for the 654 sites was three);
2. Occurred in areas above 200m elevation where agricultural pressures were not expected to impact on water-bodies;
3. Occurred in areas where field work was deemed unfeasible due to geographical constraints (Roberts, 2014).

This resulted in a net total of 356 sample point sites, from which catchments draining into these sample points were delineated using the Hydrology tool-set, in Arc-Map GIS ver. 10.1 and a national DEM (20m).

Out of the 356 study sites, 174 sites that had Maintained, Lost or Gained in status during the period 2004-2014, were identified to assess land cover change between the period 2006-2012 (Figure 2.1a). The years 2004 and 2005, and 2013 and 2014 were included to ensure correct listing of status, as EPA Q-value monitoring is carried out once every three years. These 174 sites are broken down as:

- 75 sites that had consistently Maintained high status (2004-2014)
- 35 sites that had Lost status (going from high in the 2004-2006 monitoring period to below high during 2010 - 2014)
- 64 sites that Gained in status (below high in 2004-2006 to high in 2010-2014)

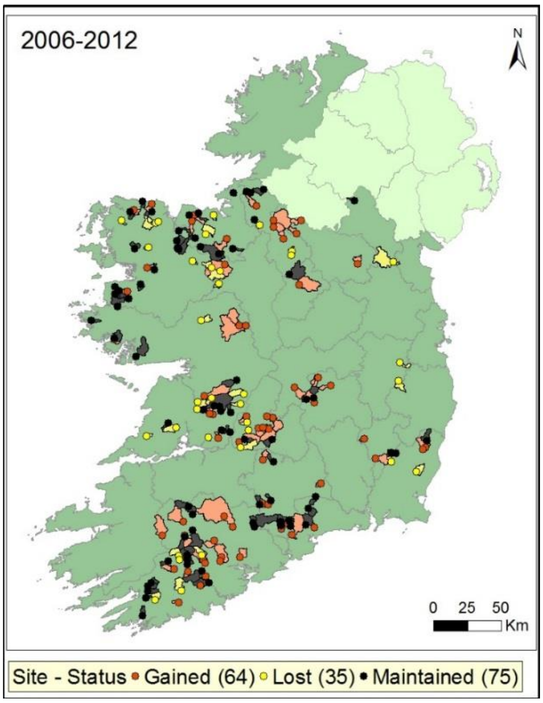
Additionally, to assess land cover change between the period 2000-2006 172 sites were identified (Figure 2.1b) from the original 356 sites and are broken down as:

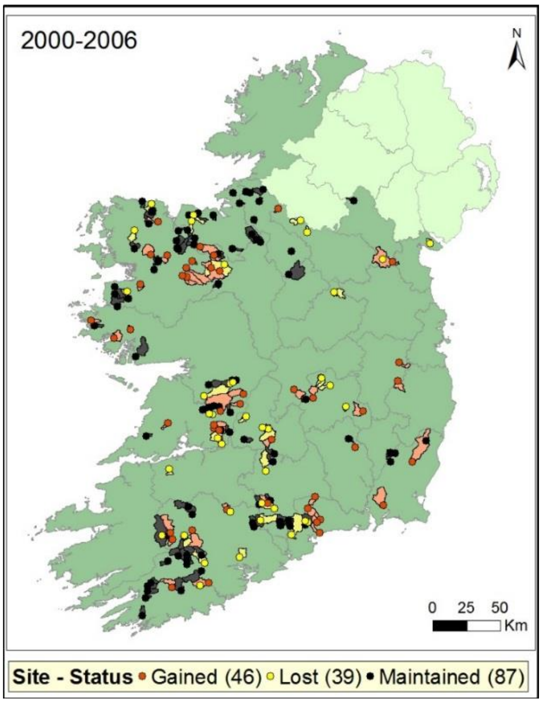
- 87 sites that had consistently Maintained high status (1998-2008)
- 39 sites that had Lost status (going from high in the 1998-2000 monitoring period to below high during 2005, 2006, 2007 or 2008)
- 46 sites that Gained in status (below high in 1998-2000 to high in 2005, 2006, 2007 or 2008).

Again, the years 1998 and 1999, and 2007 and 2008 were included to ensure correct listing of status.

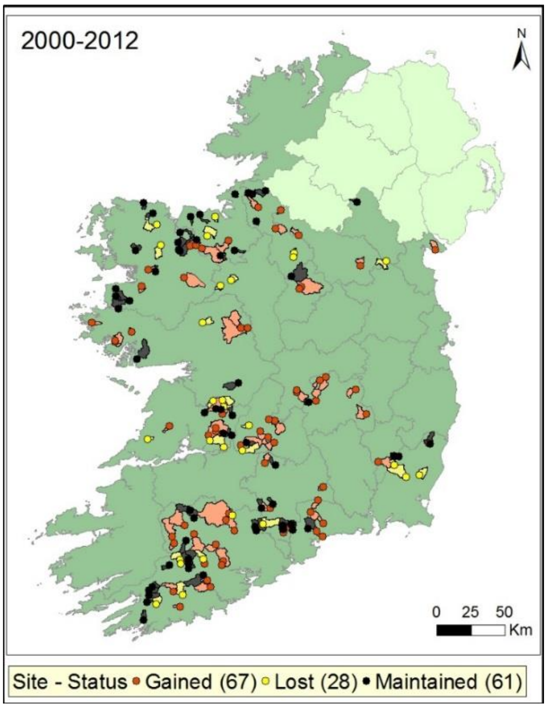
Furthermore, to assess land cover change between the period 2000-2012, 156 sites were identified (Figure 2.1c) from the 356 sites and are broken down as:

- 61 sites that had consistently Maintained high status (1998-2014)
- 28 sites that had Lost status (going from high in the 1998-2000 monitoring period to below high during 2010 - 2014).
- 67 sites that Gained in status (below high in 1998-2000 to high in 2010-2014). Again, the years 1998 and 1999, and 2013 and 2014 were included to ensure correct listing of status.





a) **b)**



c)

Figure 2.1. Map of Ireland showing the location of high status sites and their associated watersheds, for each Lost, Maintained and Gained category, for the time period: a) 2006-2012; b) 2000-2006; and c) 2000-2012.

2.2.2. Catchment attributes

The catchment areas of sites identified as suitable for land cover change assessment were used to clip CORINE (Coordination of Information on the Environment) land cover (CLC) layers for Ireland (http://gis.epa.ie/), in order to isolate the land cover types through which water had drained (Figure 2.2). The CLC datasets are part of the wider CORINE project that was set up by the European Commission with the aim of harmonising the compilation of environmental data across Member States, especially through use of geographical information systems (EEA technical report, 1995). The CLC dataset classifies land at three hierarchical levels. The first level consists of five main land cover layers, namely: Artificial Surfaces, Agricultural areas, Forest and Semi-natural areas, Wetlands, and Water Bodies (EEA technical report, 1995). The second and third levels further sub-divide these Level 1 classifications into 15 and 44 additional categories, respectively. For the purposes of this study, the Level 2 classification was selected, to reduce data processing, and because several Level 2 classifications had only single corresponding Level 3 classifications. The first CLC map was finalised in 1990 with updates occurring in 2000, 2006, and 2012. CORINE land cover layers have a minimum mapping unit of 25 hectares (EEA technical report, 1995).

To assess land cover change between the period 2006-2012, the CLC layers for the years 2006 and 2012, were clipped with the watersheds of sites identified in section 2.2.1. “Study Sites” (Figure 2.2). From these clipped CLC/watershed layers, for each individual year (2006 and 2012), the area and number of patches, for each land-cover type (e.g. Arable Land, Pastures, etc.) within each status category (Gained, Lost, Maintained) were determined.

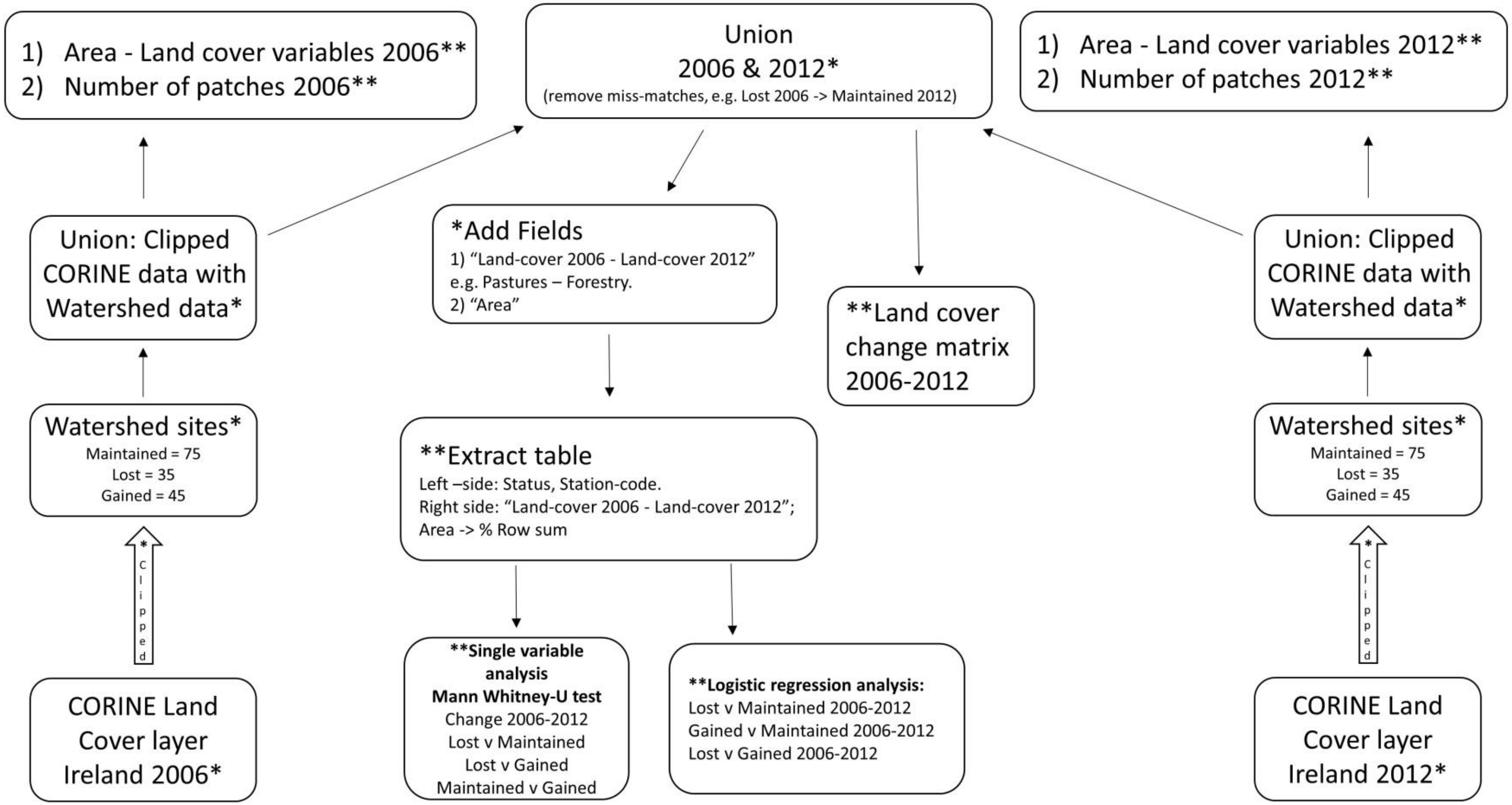


Figure 2.2. Flow chart of methods used for generating and analysing land cover change variables (2006-2012); for Lost, Maintained and Gained status categories. Note: * ArcMap GIS used; ** SPSS used. This method was repeated for the 2000-2006 and 2000-2012 land cover change periods.

The change in area for each land cover variable for the years 2006-2012 was calculated by subtracting the area for each 2006 land cover variable from the 2012 land cover variables. This was then normalised as a percentage. Additionally, the overall percentage change for each land cover variable for the period 2006-2012 was calculated as:

$$\% \, Overall \; area \; change = \frac{(area \; 2012 - area \; 2006)}{\sum(area \; 2012 - area 2006)} * 100 \qquad \text{[Eq. 2.1.]}$$

These steps were repeated for the land cover change periods 2000-2006, and 2000-2012, using the CLC layers for 2000 and 2006, and 2000 and 2012, respectively.

Separately, the Union function in GIS was used to combine clipped CLC/watershed data from individual years, e.g. 2006 and 2012, to generate land cover change matrices; for example, to record the proportion of Arable land that changed to Forest, or Arable land that remained as Arable, between the years 2006-2012 (Figure 2.2). This also enabled, the proportion of land that had changed from one land cover type to another, for each individual Lost, Maintained, and Gained site to be determined. This was again repeated for the periods 2000-2006, 2000-2012. As some watersheds are shared between more than one sample site, this generated some mismatches in the Union generated datasets. These mismatches were removed.

2.2.3. Data analysis

Using the Union generated data-sets for the periods 2006-2012, 2000-2006 and 2000-2012, the difference between Lost and Maintained, Lost and Gained, and Gained and Maintained, based on the proportional distribution of land cover variables (i.e. the proportion of land that stayed the same or that changed from one land cover type to another within each catchment), was determined. This was conducted using non-parametric Mann-Whitney U tests as the catchments are independent of each other. Again using the Union generated data-sets, the land cover variables that were the strongest predictor of change between Lost and Maintained status, Maintained and Gained, and Gained and Lost, were determined using logistic regression analysis. For logistic regression analysis the Forward Conditional method was employed.

As slope and alkalinity are likely factors in the distribution of land cover types these were included in the logistic regression model. These slope and alkalinity values are based on the model described by Kelly-Quinn et al. (2005), and were determined from RIVtype values assigned by the EPA for each site (EPA, 2007 - http://www.epa.ie/pubs/reports/water/other/wfd/#.VnbVG1IVHL9 and Bryan Kennedy (EPA, personal communication). Both Mann-Whitney U and logistic regression analysis were carried out in SPSS version 22 (IBM, 2013).

2.3. Results

2.3.1. Land cover change

An example of land cover change between the period 2000 to 2012 for a single site (30N010100) is presented in Figure 2.3, and shows, for example, the loss of Arable Land and Inland Wetlands during this period, and increases in land classified as Pastures. Additionally, when considering the land cover change results, there are two components to consider. First, the percentage change in the amount of a land cover type from one period to the next. So for example, if Urban Fabric was 10 hectares in 2006 and 20 hectares in 2012, this is an increase of 100 %. However, if Pastures was 2000 hectares in 2006 and increased to 2500 hectares (an increase of 500 hectares) by 2012, this is an increase of only 25 %. The second component therefore, as highlighted by the Pastures example, is the requirement to consider the amount of land cover change relative to the "overall land cover change". So in the example, the 10 hectares increase in Urban Fabric is minimal compared to the 500 hectares increase in Pastures.

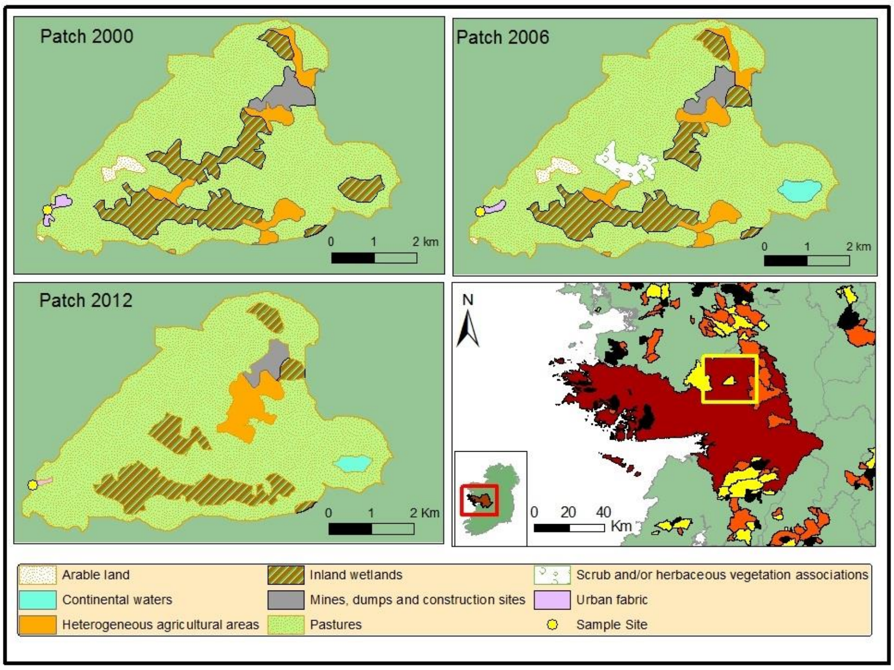


Figure 2.3. Land cover change between the period 2000 to 2012 for the site 30N010100.

2.3.2. Land cover period 2006-2012

For the sites assessed between the 2006 and 2012 period, Level 2 land cover change occurred in 23.7 %, 25.3 %, and 25.2 % of the land within the Gained, Lost and Maintained watersheds, respectively. There was an increase in land cover occurring as Forest; Mines, Dumps and Construction; and Pastures; for each of the Lost, Maintained and Gained status categories (Table 2.1 and Figure 2.4a). The land cover type with the largest "percentage" increase between 2006-2012 for Gained and Maintained was Open Spaces with little/no veg., increasing by 229 % and 118.2 % of its amount in 2006, respectively. Urban Fabric (29.7 %) and Forest (28.1 %) had the largest percentage increase for land occurring in Lost catchments. Declines in the amount of land recorded as Arable land, Continental waters, Inland wetlands, and Scrub, for each status classification was observed. A comparison of each status category revealed that catchments that Maintained status retained more of its land as Inland Wetlands, than Lost or Gained catchments, but had a greater reduction in land classified as Arable Land (Table 2.1 and Figure 2.4a). While each status gained some Pasture and Forestry, this was greater in the Lost category for Forest, and least in the Lost category for Pasture. Lost sites additionally, lost more Inland Wetlands than the other two categories, and gained more Urban Fabric.

Table 2.1. Change in the no. of patches, area (Ha), percent area, and percent overall change between the years 2006-2012 for each land type. *land cover type not present before 2006.

	Change 2006-2012											
	Gained				Lost				Maintained			
Class	**No. Patches**	**Area (Ha)**	**% Area**	**% Overall**	**No. Patches**	**Area (Ha)**	**% Area**	**% Overall**	**No. Patches**	**Area (Ha)**	**% Area**	**% Overall**
Arable land	-158	-10743.78	-56.44	-23.66	-17	-1325.63	-49.04	-10.23	-61	-2949.98	-86.15	-11.16
Artificial non-agric. veg. areas	0	4.57	3.37	0.01	2	26.66	*	0.21	0	-0.03	-0.53	0.00
Continental waters	0	-17.35	-3.13	-0.04	-2	-80.72	-15.60	-0.62	-2	-177.42	-9.19	-0.67
Forest	31	4718.29	17.94	10.39	15	2333.90	28.09	18.01	40	3552.72	13.39	13.44
Heterogeneous agric. areas	-121	-852.87	-3.78	-1.88	-35	1362.27	12.91	10.51	-23	-932.68	-5.78	-3.53
Ind., comm. and trans. Units	0	0.63	0.69	0.00	1	1.71	1.46	0.01				
Inland wetlands	-15	-5172.79	-9.89	-11.39	-28	-3425.09	-16.52	-26.44	-33	-5850.83	-8.60	-22.13
Mines, dumps and constr. sites	0	28.50	9.03	0.06	1	4.21	1.76	0.03	3	41.04	52.58	0.16
Open spaces with little/no veg.	1	14.40	229.04	0.03	2	39.98	*	0.31	3	856.15	118.16	3.24
Pastures	-69	17938.05	11.36	39.50	-27	2653.40	6.23	20.48	-25	8764.93	11.33	33.16
Scrub and/or herb. veg. assoc	-143	-5920.83	-15.84	-13.04	-50	-1646.41	-12.42	-12.71	-120	-3305.99	-9.76	-12.51
Urban fabric	0	3.18	0.51	0.01	1	55.72	29.70	0.43	0	2.09	5.60	0.01

Table 2.2. Change in the no. of patches, area (Ha), percent. area, and percent. overall change between the years 2000-2006 for each land type.

	Change 2000-2006											
	Gained				Lost				Maintained			
Class	**No. Patches**	**Area (Ha)**	**% Area**	**% Overall**	**No. Patches**	**Area (Ha)**	**% Area**	**% Overall**	**No. Patches**	**Area (Ha)**	**% Area**	**% Overall**
Arable land	-8	-134.20	-1.08	-1.24	-5	-438.91	-9.44	-3.84	-2	-58.82	-2.58	-0.37
Artificial non-agric. veg. areas	0	0.00	0.00	0.00	0	1.65	2.76	0.01	0	1.65	35.70	0.01
Continental waters	0	7.04	0.60	0.07	-1	-140.16	-19.56	-1.22	-5	-120.55	-6.72	-0.76
Forest	13	-256.55	-1.49	-2.38	-18	-760.52	-5.34	-6.65	24	-2633.09	-9.61	-16.61
Heterogeneous agric. areas	8	2182.26	15.07	20.24	-5	216.50	3.24	1.89	13	437.46	2.24	2.76
Ind., comm. and trans. Units	0	3.81	3.35	0.04	0	-0.02	-0.05	0.00				0
Inland wetlands	41	-3175.28	-7.87	-29.45	38	-3971.71	-12.66	-34.71	14	-3707.67	-5.33	-23.39
Mines, dumps and constr. sites	1	6.64	3.53	0.06	-4	-77.89	-28.25	-0.68	-2	-74.73	-41.71	-0.47
Open spaces with little/no veg.	-1	-89.79	-99.93	-0.83	0	0.00	0.00	0.00	0	-104.13	-12.62	-0.66
Pastures	3	-1665.66	-2.02	-15.45	4	-332.32	-0.45	-2.90	-1	-1225.25	-1.44	-7.73
Scrub and/or herb. veg. assoc	58	3191.50	14.02	29.60	66	5415.02	33.56	47.32	88	7447.78	28.86	46.99
Urban fabric	1	-69.78	-12.70	-0.65	1	88.36	38.68	0.77	2	37.36	39.86	0.24

Table 2.3. Change in the no. of patches, area (Ha), percentage area, and percentage overall change between the years 2000-2012 for each land type. * indicates land cover type not present in before 2000.

	Change 2000-2012											
	Gained				Lost				Maintained			
Class	**No. Patches**	**Area (Ha)**	**% Area**	**% Overall**	**No. Patches**	**Area (Ha)**	**% Area**	**% Overall**	**No. Patches**	**Area (Ha)**	**% Area**	**% Overall**
Arable land	-122	-10363.84	-60.92	-27.30	-13	-1841.95	-30.44	-17.97	-44	-1531.64	-88.64	-8.22
Artificial non-agric. veg. areas	0	75.55	53.74	0.20	2	30.88	*	0.30	0	1.62	34.98	0.01
Continental waters	-5	-244.31	-18.94	-0.64	1	2.76	0.67	0.03	0	-80.69	-6.53	-0.43
Forest	121	2735.27	9.27	7.21	9	2230.78	26.17	21.76	47	-283.21	-1.45	-1.52
Heterogeneous agric. areas	-75	-839.07	-3.73	-2.21	-16	654.54	7.22	6.39	-10	561.76	5.08	3.01
Ind., comm. and trans. Units	1	68.51	153.64	0.18								
Inland wetlands	6	-7532.39	-14.90	-19.84	-25	-3261.17	-18.03	-31.82	-11	-7424.65	-13.80	-39.83
Mines, dumps and constr. sites	1	9.48	4.59	0.02	0	-22.03	-15.87	-0.21	3	30.49	34.41	0.16
Open spaces with little/no veg.	1	211.33	235.35	0.56	0	53.92	60.05	0.53	3	551.17	66.81	2.96
Pastures	-50	14695.17	8.84	38.71	-12	1927.51	5.42	18.80	-14	4048.27	6.91	21.72
Scrub and/or herb. veg. assoc	-74	1015.11	3.01	2.67	-4	155.62	1.64	1.52	-26	4087.44	23.13	21.93
Urban fabric	6	169.19	34.77	0.45	2	69.15	40.82	0.67	2	39.45	*	0.21

a)

b)

c)

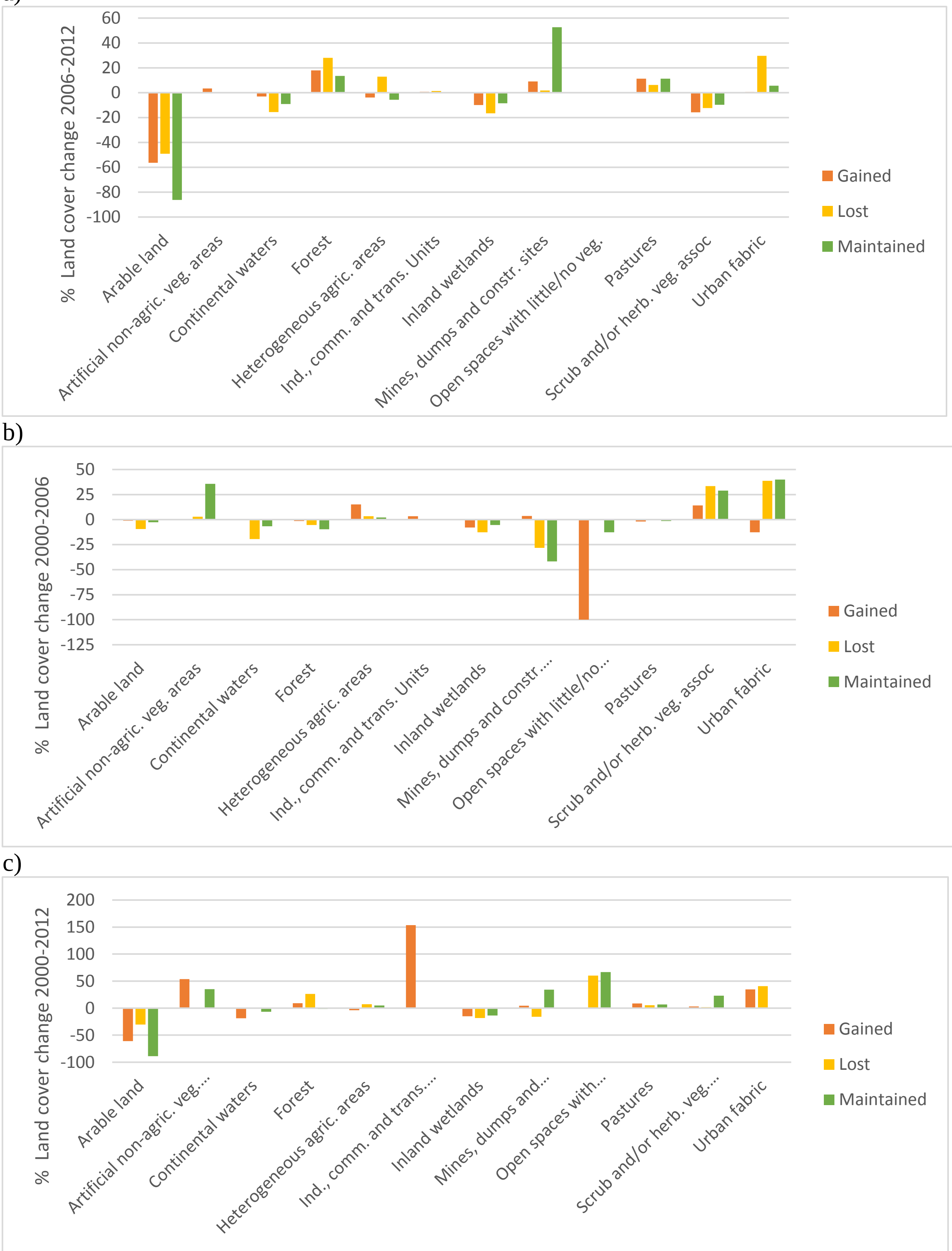
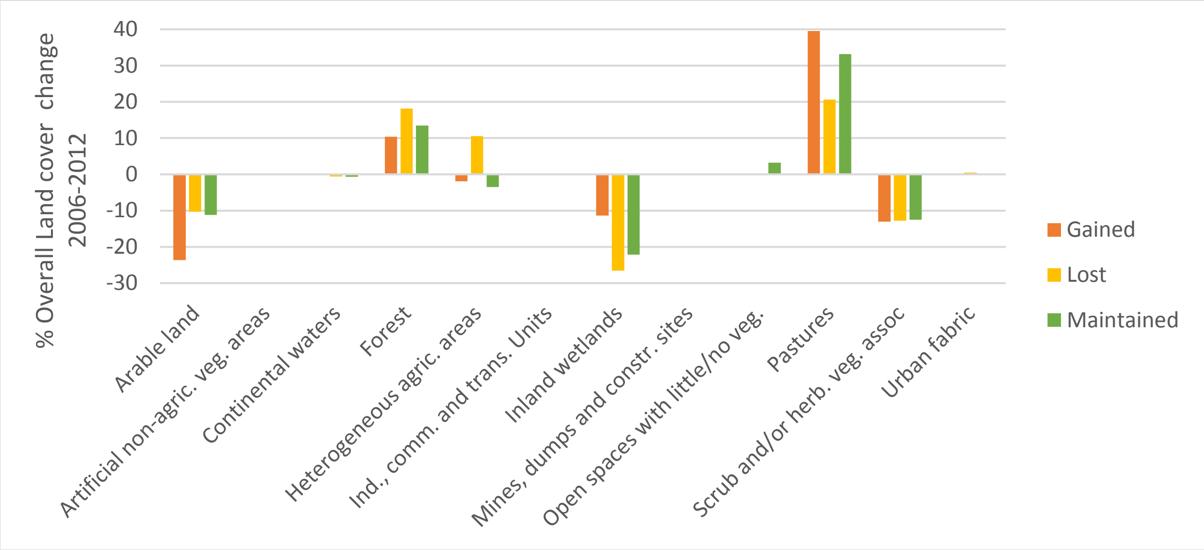
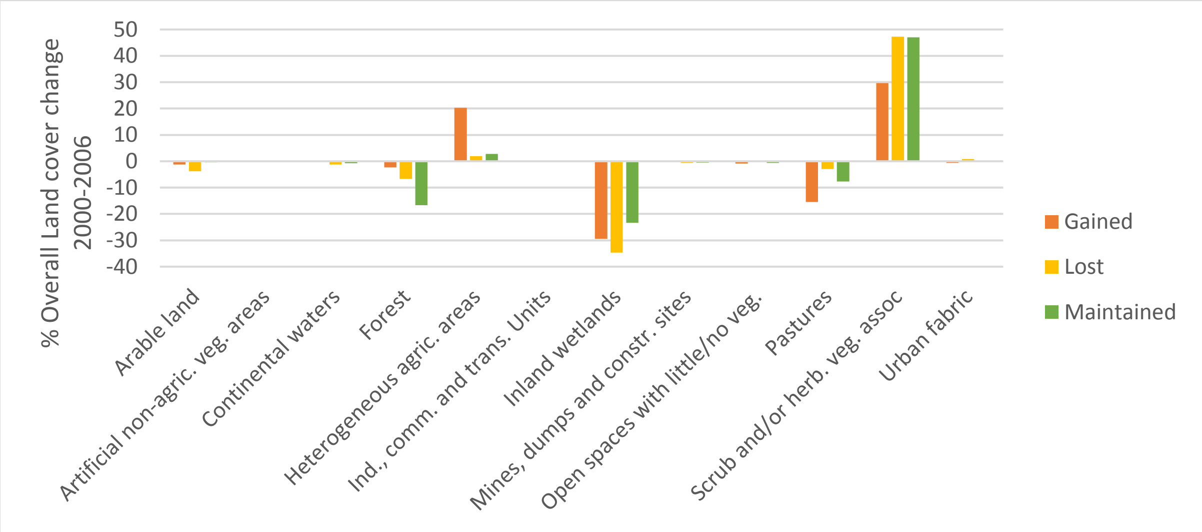


Figure 2.4. The percentage change for each land cover variable (not normalised as a percentage of the overall land cover change) for Lost, Maintained, and Gained, for: a) 2006-2012; b) 2000-2006; and c) 2000-2012. Note: Open spaces with little/no veg. values not included on graph a (Gained and Maintained) or graph c (Gained) because of scaling (See Tables 1 and 3).

a)



b)



c)

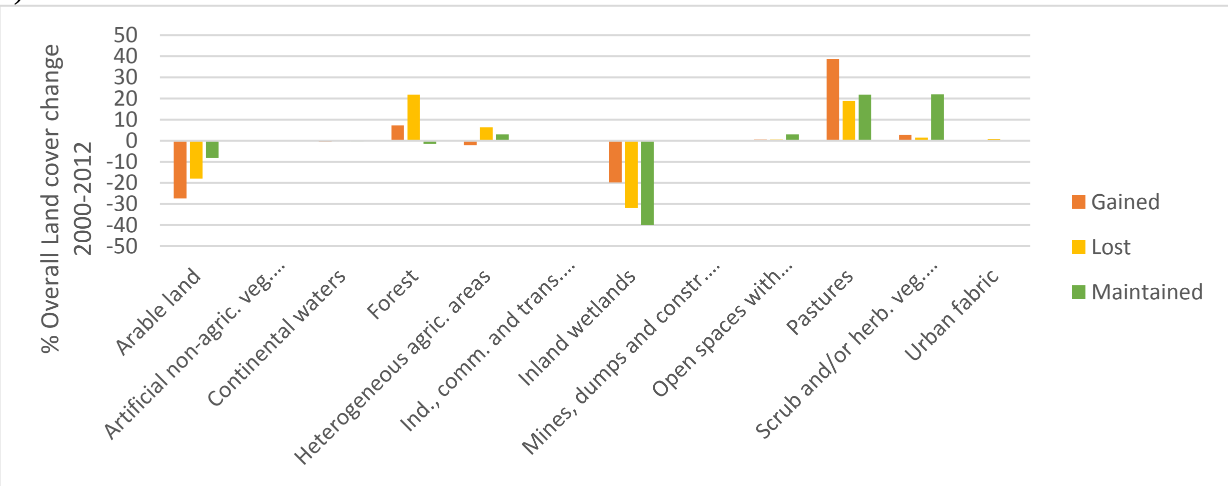


Figure 2.5. The change for each land cover variable as a proportion of overall change for Lost, Maintained, and Gained, for: a) 2006-2012; b) 2000-2006; and c) 2000-2012.

However, when expressed as a proportion of the overall land cover change for each Lost, Maintained and Gained categories (Figure 2.5a), the largest gains were for Pasture and Forestry, while Arable land, Inland Wetlands and Scrub had the largest losses. In contrast to the individual land cover changes reported in Table 2.1 and Figure 2.4a, the overall land cover (Table 2.1 and Figure 2.5a), indicated that Maintained, lost a greater proportion of Inland Wetlands and a lesser proportion of Arable land than Gained, and gained more Forest. Five of the Gained, and six of the Lost and Maintained land cover variables recorded a decrease in the number of patches, with all other variables, either recording an increase or not changing (Table 2.1).

The percentage and direction of change for each land cover type (2006-2012) that make up the Gained, Lost and Maintained watersheds (combined), is presented in Table 2.4. For example, Table 2.4 shows that the percentage of land that was Inland Wetlands in 2006 and that remained as Inland Wetlands in 2012, was 13.7 %, 14.9 % and 24.3 % for the Gained, Lost and Maintained status categories, respectively. The percentage of land that was Forest in 2006 and that changed to Pastures in 2012 was 0.5 %, 0.3 % and 0.5 % for each of the Gained, Lost and Maintained status categories, respectively.

Based on the proportional distribution of land cover variables, Mann-Whitney U tests revealed significant differences between Lost and Maintained for sixteen land cover change variables (e.g. changing from Arable land to Forestry), nine variables for Lost against Gained, and twenty-seven variables for Gained against Maintained (see supplementary data – Appendix A). Additionally, there was a significant difference between the “slope” (RIVtype classification) values of both Maintained and Lost, and

Maintained and Gained. The logistic regression model between Maintained and Lost was significant (chi square = 39.51, $p < 0.001$), explaining 42.3 % (Nagelkerke R^2) of the variation in status, with an overall prediction success of 77.3 % (90.7 % Maintained, 48.6 % Lost).

Table 2.4. Percentage and direction of change (2006-2012) for each land cover type that make up the Gained, Lost and Maintained watersheds (combined). Note: only values greater than 0.5 % in at least one status group are displayed.

Land cover 2006 - Land cover 2012	Gained %	Lost %	Maintained %
Continental waters - Continental waters	0.2	0.4	0.7
Heterogeneous agricultural areas - Scrub and/or herbaceous vegetation associations	0.3	0.5	0.3
Inland wetlands - Forest	0.4	0.7	0.6
Forest - Pastures	0.5	0.3	0.5
Pastures - Arable land	0.5	0.3	0.1
Inland wetlands - Pastures	0.6	1.8	1.1
Pastures - Forest	0.7	0.4	0.6
Scrub and/or herbaceous vegetation associations - Inland wetlands	0.7	0.9	1.9
Inland wetlands - Scrub and/or herbaceous vegetation associations	0.8	2.0	2.5
Pastures - Scrub and/or herbaceous vegetation associations	0.9	0.5	0.7
Inland wetlands - Heterogeneous agricultural areas	1.0	1.4	0.9
Scrub and/or herbaceous vegetation associations - Heterogeneous agricultural areas	1.1	1.3	1.0
Scrub and/or herbaceous vegetation associations - Pastures	1.2	0.9	1.4
Forest - Scrub and/or herbaceous vegetation associations	1.7	1.9	2.1
Pastures - Heterogeneous agricultural areas	1.8	2.5	1.9
Arable land - Arable land	2.0	1.0	0.1
Scrub and/or herbaceous vegetation associations - Forest	2.6	3.4	2.9
Heterogeneous agricultural areas - Heterogeneous agricultural areas	2.6	6.5	2.8
Heterogeneous agricultural areas - Pastures	3.7	2.1	3.3
Arable land - Pastures	3.8	1.7	1.3
Forest - Forest	5.8	5.8	8.7
Scrub and/or herbaceous vegetation associations - Scrub and/or herbaceous vegetation associations	6.2	6.8	7.6
Inland wetlands - Inland wetlands	13.7	14.9	24.3
Pastures - Pastures	45.6	38.7	30.2
Total	**98.4**	**97.0**	**97.6**

Forestry-Forestry ($p<0.002$), Forest-Heterogeneous Agricultural areas ($p= 0.037$) and Inland wetlands–Inland wetlands ($p<0.001$) were identified as significant predictors of difference in status. Land that changed from Forestry to Heterogeneous Agricultural areas was 17.5 times more likely to result in Lost status, whereas land that remained as Forestry or Inland Wetlands reduced chance of Lost status occurring by 15 % and 4 %, respectively. The addition of typology (hardness and slope) did not improve the model, with slope bringing the Nagelkerke R^2 value down to 27.9 %.

The logistic regression model between Maintained and Gained was significant (chi square = 26.11, $p < 0.001$), explaining 22.9 % (Nagelkerke R^2) of the variation in status, with an overall prediction success of 68.3 % (73.3 % Maintained, 62.5 % Gained). Pastures-Pastures ($p<0.001$) and Scrub-Forest ($p= 0.045$) were identified as significant predictors of difference in status. Land that remained as Pastures and land that changed from Scrub to Forestry was 1.04 and 1.12 times more likely to be identified as Gained status, respectively, as opposed to Maintained status. The addition of typology (hardness and slope) improved the model, explaining 34.5 % (Nagelkerke R^2) of the variation in status, with an overall prediction success of 71.2 % (78.7 % Maintained, 62.5 % Gained) and chi square = 41.56, $p < 0.001$). Pastures-Pastures ($p<0.001$) was again identified as a significant predictor of difference in status, again increasing the likelihood of Gained occurring by 1.04 times. Additionally, Forest-Inland Wetlands ($p=0.039$) was identified as increasing the likelihood of Gained status occurring by 2.46 times, while sites occurring in Hardness Cat. 2 typology ($p=0.005$) reduced the likelihood of sites being identified as Gained.

The logistic regression model between Gained and Lost was significant (chi square = 33.06, $p < 0.001$), explaining 39 % (Nagelkerke R^2) of the variation in status, with an overall prediction success of 76.8 % (93.8 % Gained, 45.7 % Lost). Heterogeneous Agric. – Inland Wetlands ($p=0.04$), Inland Wetlands – Pastures ($p= 0.022$), and Scrub

– Heterogeneous Agric. (p=0.012), increased the likelihood of Lost status being identified by 2.08, 1.25, and 1.3 times respectively. Heterogeneous Agric. – Pastures (p=0.017) reduced the likelihood of Lost status being identified by 21 %. The addition of typology (hardness and slope) did not change the model, again explaining 34.5 % (Nagelkerke R^2) of the variation in status, with an overall prediction success of 76.8 % and the same predictor variables being identified.

2.3.3. Land cover period 2000-2006

For the sites assessed between the 2000 and 2006 period, Level 2 land cover change occurred in 10.3 %, 10.1 % and 10.5 % of the land within the Gained, Lost and Maintained watersheds, respectively. There was an increase in land cover occurring as Scrub for each of the Lost, Maintained and Gained status categories (Table 2.2 and Figure 2.4b), while there was a decrease in Arable Land, Forest, Inland Wetlands, and Pastures for all status categories. For both Lost and Maintained, Urban Fabric had the largest percentage increase between years (38.7 % and 39.9 % respectively). Heterogeneous Agric. Areas (15.1 %) and Scrub (14 %) had the largest percentage increase between years for Gained. Open Spaces with little/no Veg. had the largest percentage decline for Gained (-99.9 %), while Mines, Dumps and Construction had the largest declines between years for Lost (-28.3 %) and Maintained (-41.7 %). In contrast to Lost and Maintained sites, Gained sites lost Urban Fabric, but gained Mines, dumps and Construction. As with 2006-2012, Maintained status lost less Inland Wetlands, than Lost or Gained (Table 2.2 and Figure 2.4b).

When expressed as a proportion of the overall land cover change for each Lost, Maintained and Gained categories (Figure 2.5b), the largest gains were for Scrub, while Inland Wetlands had the largest losses. The overall land cover figure (Table 2.2 and Figure 2.5b) indicated that Maintained sites, lost a lesser proportion of Inland Wetlands, than Lost or Gained. Maintained, however, lost a greater proportion of Forest than Lost or Gained, and a greater proportion of Pastures than Lost. Two Gained, five Lost and four Maintained land cover variables recorded a decrease in the number of patches, with all other variables, either recording an increase or not changing (Table 2.2).

The percentage and direction of change for each land cover type (2000-2006) that make up the Gained, Lost and Maintained watersheds (combined), is presented in Table 2.5. Mann-Whitney U tests revealed significant differences between Lost and Maintained for eight land cover change variables (e.g. changing from Arable land to Forestry), one variable for Lost against Gained, and nine variables for Gained against Maintained (see supplementary data in Appendix A). Additionally, there was a significant difference between the hardness (alkalinity) (RIVtype classification) values of Maintained and Lost, and the slope values of Maintained and Gained. The logistic regression model between Maintained and Lost was significant (chi square = 16.22, $p < 0.001$), explaining 17 % (Nagelkerke R^2) of the variation in status, with an overall prediction success of 73.8 % (93.1 % Maintained, 30.8 % Lost). Pastures-Pastures ($p<0.001$), were identified as a significant predictor of difference in status, increasing the chance of Lost being identified by 1.03 times. The addition of typology (hardness and slope) improved the model, explaining 30 % (Nagelkerke R^2) of the variation in status, with an overall prediction successes of 73 % (87.4 % Maintained, 41 % Lost) and chi square = 30.23, $p < 0.001$).

Table 2.5. Percentage and direction of change (2000-2006) for each land cover type that make up the Gained, Lost and Maintained watersheds (combined). Note: only values greater than 0.5 % in at least one status group are displayed.

Land cover 2000 - Land cover 2006	Gained %	Lost %	Maintained %
Scrub and/or herbaceous vegetation associations - Inland wetlands	.5%	.3%	.5%
Scrub and/or herbaceous vegetation associations - Pastures	.5%	.4%	.2%
Continental waters - Continental waters	.5%	.4%	.6%
Pastures - Scrub and/or herbaceous vegetation associations	.7%	.8%	.5%
Scrub and/or herbaceous vegetation associations - Heterogeneous agricultural areas	.8%	.1%	.2%
Pastures - Heterogeneous agricultural areas	.8%	.1%	.5%
Scrub and/or herbaceous vegetation associations - Forest	1.2%	1.3%	1.1%
Inland wetlands - Scrub and/or herbaceous vegetation associations	1.7%	2.6%	1.6%
Forest - Scrub and/or herbaceous vegetation associations	1.7%	2.2%	2.7%
Arable land - Arable land	6.3%	2.8%	.9%
Heterogeneous agricultural areas - Heterogeneous agricultural areas	6.9%	4.2%	7.6%
Forest - Forest	7.0%	7.1%	8.6%
Scrub and/or herbaceous vegetation associations - Scrub and/or herbaceous vegetation associations	8.9%	8.7%	9.0%
Inland wetlands - Inland wetlands	18.6%	17.7%	27.4%
Pastures - Pastures	41.0%	48.6%	34.9%
Total	**97.3%**	**97.3%**	**96.5%**

Pastures-Pastures ($p<0.001$) was again identified as a significant predictor of difference in status, again increasing the likelihood of Lost occurring by 1.04 times. Additionally, Scrub-Forest ($p=0.029$) was identified as increasing the likelihood of Lost status occurring by 1.19 times, while sites occurring in Hardness category 1 ($p=0.003$) reduce the likelihood of sites being identified as Lost by 84 %.

The logistic regression model between Maintained and Gained was significant (chi square = 26.50, $p < 0.001$), explaining 24.9 % (Nagelkerke R^2) of the variation in status, with an overall prediction success of 73.7 % (96.6 % Maintained, 30.4 % Gained). Arable Land-Arable Land ($p<0.037$), were identified as a significant predictor of difference in status, increasing the chance of Gained being identified by 1.17 times. The addition of typology (hardness and slope) improved the model, explaining 29.2 % (Nagelkerke R^2) of the variation in status, with an overall prediction successes of 72.2 % (90.8 % Maintained, 37 % Lost) and chi square = 31.57, $p < 0.001$). Scrub-Inland wetlands ($p=0.027$) was identified as increasing the likelihood of Gained status occurring by 1.26 times, while sites occurring in Slope Cat. 1 ($p=0.019$) increased the likelihood of sites being identified as Gained 29.60 times. The logistic regression model between Lost and Gained did not determine any significant differences.

2.3.4. Land cover period 2000-2012

For the sites assessed over the full period between the 2000 and 2012 period, Level 2 land cover change occurred in 29.4%, 31.6%, and 28.3 % of the land within the Gained, Lost and Maintained watersheds, respectively. There was an increase in land cover occurring as Open Spaces with little/no Veg., Pastures and Scrub for each of the Lost, Maintained and Gained status categories (Table 2.3 and Figure 2.4c), while Forest and Urban Fabric increased in the Lost and Gained categories only. Arable land and Inland wetlands declined for all status categories. Open Spaces with little/no Veg. had the largest percentage increase between years for Gained (235.35 %), Lost (60.05 %), and Maintained (66.81 %); while Arable Land had the largest declines: -60.92 %, -30.44 % and -88.64 % respectively. A comparison of each status revealed that Maintained status lost less Inland Wetlands, than Lost or Gained, but lost more Arable Land (Table 2.3 and Figure 2.4c), while increases in Pasture and Scrub were highest in the Gained and Maintained categories respectively.

When expressed as a proportion of the overall land cover change for each Lost, Maintained and Gained categories however (Figure 2.5c), the largest gains were for Pastures, while Arable land and Inland Wetlands had the largest losses. In contrast to the individual land cover changes reported in Table 2.3 and Figure 2.4c, the overall land cover (Table 2.3 and Figure 2.5c) indicates that Maintained lost a greater proportion of Inland Wetlands, than Lost or Gained, and a lesser proportion of Arable Land. Pastures increased more in Maintained and Gained categories than in Lost, while Forestry increased more in the Lost category. Five of the Gained, Lost and Maintained land cover variables recorded a decrease in the number of patches, with all other variables, either recording an increase or not changing (Table 2.3).

The percentage and direction of change for each land cover type (2000-2012) that make up the Gained, Lost and Maintained watersheds (combined), is presented in Table 2.6. Mann-Whitney U tests revealed significant differences between Lost and Maintained for ten land cover change variables (e.g. changing from Arable land to Forestry), three variables for Lost against Gained, and sixteen variables for Gained against Maintained (see supplementary data in Appendix A).

The logistic regression model between Maintained and Lost was significant (chi square = 14.24, $p < 0.001$), explaining 20.8 % (Nagelkerke R^2) of the variation in status, with an overall prediction success of 71.9 % (88.5 % Maintained, 35.7 % Lost). Inland Wetlands – Inland Wetlands ($p=0.002$), were identified as a significant predictor of difference in status, reducing the chance of Lost being identified by 4.1 %. The addition of typology improved the model, explaining 36.2 % (Nagelkerke R^2) of the variation in status, with an overall prediction successes of 78.7 % (88.5 % Maintained, 57.1 % Lost) and chi square = 26.53, $p < 0.001$). Inland Wetlands – Inland Wetlands ($p=0.026$) was again identified as a significant predictor of difference in status, again reducing the likelihood Lost occurring by 3.1 %.

Table 2.6. Percentage and direction of change (2000-2012) for each land cover type that make up the Gained, Lost and Maintained watersheds (combined). Note: only values greater than 0.5 % in at least one status group are displayed.

Land cover 2000 - Land cover 2012	Gained %	Lost %	Maintained %
Inland wetlands - Open spaces with little or no vegetation	0.1	0.0	0.5
Heterogeneous agricultural areas - Inland wetlands	0.2	0.9	0.3
Continental waters - Continental waters	0.3	0.4	0.6
Heterogeneous agricultural areas - Scrub and/or herbaceous vegetation associations	0.4	0.5	0.4
Pastures - Arable land	0.4	1.2	0.0
Forest - Pastures	0.4	0.5	0.6
Inland wetlands - Heterogeneous agricultural areas	0.5	1.4	1.0
Inland wetlands - Forest	0.9	1.2	1.1
Inland wetlands - Pastures	1.1	0.9	1.4
Pastures - Scrub and/or herbaceous vegetation associations	1.1	1.0	1.0
Scrub and/or herbaceous vegetation associations - Heterogeneous agricultural areas	1.1	1.0	1.1
Pastures - Forest	1.2	0.8	1.1
Scrub and/or herbaceous vegetation associations - Inland wetlands	1.2	1.2	1.8
Scrub and/or herbaceous vegetation associations - Pastures	1.5	0.7	1.3
Arable land - Arable land	1.6	3.5	0.1
Inland wetlands - Scrub and/or herbaceous vegetation associations	1.8	2.7	3.4
Heterogeneous agricultural areas - Heterogeneous agricultural areas	2.1	5.7	2.9
Scrub and/or herbaceous vegetation associations - Forest	2.5	3.8	2.1
Pastures - Heterogeneous agricultural areas	2.7	2.8	2.0
Forest - Scrub and/or herbaceous vegetation associations	3.3	2.9	3.8
Arable land - Pastures	3.4	3.3	0.8
Heterogeneous agricultural areas - Pastures	3.8	2.8	2.8
Scrub and/or herbaceous vegetation associations - Scrub and/or herbaceous vegetation associations	4.1	3.8	4.4
Forest - Forest	5.1	5.8	7.0
Inland wetlands - Inland wetlands	11.4	14.4	25.3
Pastures - Pastures	45.9	34.4	31.1
Total	**98.1**	**98.0**	**97.9**

The logistic regression model between Maintained and Gained was significant (chi square = 26.41, $p < 0.001$), explaining 24.9 % (Nagelkerke R^2) of the variation in status, with an overall prediction success of 64.8 % (59 % Maintained, 70.1 % Gained). Forestry-Pastures (p=0.009) and Inland wetlands–Inland wetlands (p=0.019) reduced the chance of Gained status occurring by 84.3 % and 4.5% respectively, while Pastures-Pastures (p=0.028) increased the chance of Gained occurring by 1.03 times. The addition of typology improved the model, explaining 33.5 % (Nagelkerke R^2) of the variation in status, with an overall prediction success of 71.1 % (73.8 % Maintained, 68.7 % Gained) and chi square = 37.01, $p < 0.001$). Inland wetlands–Inland wetlands ($p=0.045$) reduced the chance of Gained status occurring by 1.9 %, while Arable Land-Pastures ($p=0.038$) increased the chance of Gained occurring by 1.35 times. The logistic regression model between Lost and Gained did not determine any significant differences at the $p = 0.05$ level.

Table 2.7. Summary of percentage of land cover that experienced change and the land cover type that increased and decreased most for each status category for the periods 2006-2012, 2000-2006 and 2000-2012.

Period	Status	Land cover change (%)	Overall (normalised as a percentage of the overall change)	
			Largest increase	**Largest decrease**
2006-2012	Gained	23.7	Pastures	Arable land
	Lost	23.3	Pastures, Forest	Inland wetlands
	Maintained	25.2	Pastures	Inland wetlands
2000-2006	Gained	10.3	Scrub and/or herb. veg.	Inland wetlands
	Lost	10.1	Scrub and/or herb. veg.	Inland wetlands
	Maintained	10.5	Scrub and/or herb. veg.	Inland wetlands
2000-2012	Gained	29.4	Pastures	Arable land
	Lost	31.6	Pastures, Forest	Inland wetlands
	Maintained	28.3	Pastures, Scrub and/or herb. veg.	Inland wetlands

Table 2.8. Summary of significant predictors of difference based on the logistic regression results for each status comparison for the periods 2006-2012, 2000-2006 and 2000-2012.

Period	**Status comparison**	**Logistic regression - Predictors of difference**
2006-2012	Gained vs Maintained	Pastures - Pastures; Scrub - Forest
	Lost vs Gained	Hetero. Agric. – Inland Wetlands; Inland Wetlands – Pastures; Scrub – Hetero Agric.
	Maintained vs Lost	Forestry-Forestry; Forest-Hetero. Agric. areas; Inland wetlands–Inland wetlands
2000-2006	Gained vs Maintained	Arable Land-Arable Land; Scrub-Inland wetlands (with add. of typology to model)
	Lost vs Gained	No significant difference
	Maintained vs Lost	Pastures-Pastures; Scrub-Forest
2000-2006	Gained vs Maintained	Forestry-Pastures; Inland wetlands–Inland wetlands; Pastures-Pastures;
	Lost vs Gained	No significant difference
	Maintained vs Lost	Inland Wetlands – Inland Wetlands

2.4. Discussion

Land use and land cover change has been suggested as a contributing factor to the observed declines in the number of high status sites in Ireland (Ní Chatháin et al., 2012; White et al., 2014). However, with the exception of, for example, Roberts et al. (2016), who assessed the effect land use has on the stability of high status river sites, this topic has received limited attention to date. This study therefore aimed to address this deficit in knowledge, with the primary objective of investigating the relationship between declines in high status river sites and changes in land use and land cover trends. To achieve this, land cover change assessment over three time periods: 2006-2012, 2000-2006 and 2000-2012; was conducted.

The overall findings here suggest that Maintained watersheds are staying at Maintained status because of a larger level of land associated with lower anthropogenic pressures, and that where land cover change has occurred in the 2006-2012 and 2000-2012 periods, this is somewhat associated with a change from one anthropogenically influenced land cover type to another (e.g. Arable Land to Pastures). The land cover change occurring during the 2000-2006 period, is more difficult to explain, although the amount of change within this period was less than half that of the 2006-2012 period. A major cause for concern across all status categories and all time periods is the continued loss of Inland Wetlands. As Inland Wetlands were identified as a key predictor of difference between Lost and Maintained status, perhaps they should be especially targeted to prevent further deteriorations, not least because of the potential for peatlands to sequester carbon emissions (Hooijer et al., 2010). Additionally, both blanket bog and raised bogs require protection under Annex 1 of the Habitats Directive (OJEC, 1992) and it may be useful for further studies to assess which watersheds are currently occurring in protected Special Areas of Conservation and NATURA 2000 areas. However, as land cover generally explained less than 40 % of the variation between Lost and Maintained, other factors are likely having a strong effect and this should be investigated further. One potential factor relates to the distinction between land cover and land use, especially as land cover assessments, as used in this study, do not distuinguish between the potential variations in land management practices occurring within each catchment. This is important as it may influence the status of associated water-bodies. Nevertheless, while the CORINE datasets are primarily land cover based, some elements such as artificial surfaces and agricultural areas are also discerned based on their functional attributes, and are therefore associated with land use practices (Feranec et al., 2007; Martínez-Fernández et al., 2015).

In contrast to what might be expected, the relationship between land cover and Gained status, follows more the pattern of Lost status rather than of that of Maintained. Therefore, while there is statistical reason to reject the null hypothesis (and accept that there is a relationship between declines in high status and land use and land cover trends) using the datasets available and based on Lost against Maintained Status, the relationship between Lost and Gained leaves an important caveat. This caveat again requires further investigation, not least because actions that are potentially benefiting Gained sites may be applied to Lost sites.

2.4.1. Land cover remaining the same

Maintained status sites had a higher proportion of land remaining as Inland Wetlands, Scrub, and Forest over the three land cover change periods (2006-2012, 2000-2006 and 2000-2012) (Tables 2.4, 2.5 and 2.6), while contrastingly, Lost sites had a higher proportion of land remaining as Pastures and Arable Land. Water quality impacts from Inland Wetlands and Scrub are likely to be minimal due to the relatively natural characteristics of these habitats and reduced level of anthropogenic influence. However, based on the land cover layers it is not possible to determine the proportion of Inland Wetlands that have been drained or that are being used as cut-over bog. This requires further investigation, as for example, drained peat-lands are a source of nitrogen to freshwaters (Vassiljev and Blinova, 2012), as well as potentially increasing sediment loadings (Pavey et al., 2007; Clément et al., 2009). Additionally, although the growing stage of forestry (land remaining as Forest) may have implications for water quality through acidification and the application of fertilisers (Giller and O'Halloran, 2004), potential major impacts occur during the afforestation and deforestation processes. For example, afforestation is associated with the construction of drainage channels and logging roads, which are a potential source of sedimentation (Waters, 1995; Prévost et al., 1999; Giller and O'Halloran, 2004); while early fertiliser applications on poorly absorbent peat soils, combined with the conversion from an anaerobic to aerobic environment as a result of lowering the water table, may lead to nutrient leaching to near-by water-courses (Drinan et al., 2013). Harvesting or clear-felling too, may be a considerable source of P (O'Driscoll et al., 2011), with P peaks occurring up to one year after harvesting (Rodgers et al., 2010). Again this is not reflected in the land cover layers. It should also be noted, that the CLC Level 2 classification combines plantation forestry with natural forestry, and the significance of this may require further investigation.

Several studies have demonstrated the relationship between intensive agricultural practices, and declines in sensitive invertebrate taxa, through either the impacts of nutrient enrichment (Liess et al., 2012), flow modification through drainage (Blann et al., 2009) or sedimentation (Zweig and Rabeni, 2001); and the interaction of these stressors (Quinn et al., 1997; Moss, 2008; Matthaei et al., 2010; Piggott et al., 2012; Lange et al., 2014b). Ramezani et al. (2014) for example, reported negative impacts to both invertebrates and fish following the addition of sediment to two farmland streams, and positive reactions following its removal. Sutherland et al. (2010) found a significant positive linear relationship between the percentage of fine sediment (<2 mm), embeddedness and particle mobility, and the percentage of a watershed under agricultural land, while negative relationships were observed for indicators of bed stability. Furthermore, Gillespie et al. (2014) for example, reported a significant negative relationship between freshwater invertebrate scores and the extent of flow regulation. Arable Land may be a major source of sedimentation (Waters, 1995), although Wasson et al. (2010) reported its presence to have both negative and positive impacts on invertebrate indices, with the positive impacts being related to the lack of urban pressures in agricultural basins. Hooda et al. (2000) reported that nitrate leaching from arable land, with or without the addition of manure, is typically greater than that from non-grazed grassland, although this is reversed if the grassland is intensively managed.

However, Gained status sites here also had higher levels of land remaining as Pastures and Arable Land. Additionally, for both the 2006-2012 and 2000-2012 periods the logistic regression model comparing Maintained and Lost status, highlighted land remaining as Inland Wetlands as reducing the potential for Lost status to occur, while land remaining as Pastures was identified by the 2000-2006 model as increasing the potential for Lost status to occur. Again however, logistic regression models of

Maintained v Gained, and Gained v Lost, highlighted similar patterns between Lost and Gained sites. In a comparison of urban, pasture and native forest land use, Miserendino et al. (2011) found urbanisation to have the most significant impact on nutrient levels, modifications to the physical habitat, and invertebrate communities, even though urban land was present at low levels. Here, Urban Areas too, although present at low levels, were more prominent in the watersheds of Lost and Gained Categories than that of Maintained. The findings here, with the exception of Forestry, are similar to those of Roberts et al. (2016) who, in an assessment of land use adjacent to high status water-bodies, found that agriculture (primarily grassland forage) had the greatest negative impact on water-bodies maintaining high status.

2.4.2. Land cover change

Where land cover change actually occurred according to the CORINE data, there was an increase in Pastures for each of the Gained, Lost and Maintained status categories observed here during the 2006-2012 and 2000-2012 land cover change periods, with this increase being less in the Lost category. Indeed Pastures accounted for almost 80% of the land cover type that increased within the overall Gained watersheds in the 2006-2012 and 2000-2012 periods, and for greater than 65 % (2006-2012) and 43 % (2000-2012) in the Maintained watersheds, while for Lost this was 41% (2006-2012) and 37.7 % (2000-2012). While these 2006-2012 and 2000-2012 results may imply that changes to Pastures alone are not responsible for the deterioration in status, it should be noted that the largest contributors to increases in Pastures during the 2006-2012 period were related to declines in Arable Land and Heterogeneous Agric. Areas (which are made up of a mixture of natural as well as agricultural land cover) in Gained, and Heterogenous Agric. Areas in Maintained sites, while declines in Inland Wetlands was an additionally strong factor in Lost sites. In the 2000-2012 period, the largest contributors to increased Pastures for all status categories resulted from declines in Arable Land, Heterogeneous Agric. Areas, along with Inland Wetlands and Scrub.

For the period 2006-2012, the logistic regression model also identified Forest changing to Heterogeneous Agricultural Areas as significantly increasing the chances of Lost status occurring, relative to Maintained. Given that this change from Forest (predominantly coniferous plantations in this case) to Heterogeneous Agricultural Areas accounts for less than 0.5 % of the overall watersheds for each status category, and may point to pressure hot-spots, perhaps this should be targeted to limit future deteriorations or at least warrants further investigation. In contrast, during the 2000-2006 land cover change period, there was a decline in Pastures in each status category, with Gained and Maintained status both losing more Pastures than Lost. Scrub accounted for almost 60 % of the land cover type that increased within the overall Gained watersheds in the period 2000-2006, and for greater than 90 % in the Maintained and Lost watersheds. The largest contributors to these increases for all status categories, were mainly related to declines in Forest and Inland Wetlands, and to a lesser extent Pastures, while the identification of Scrub changing to Forest as a key differentiator between Maintained and Lost may be associated with the impacts of afforestation.

2.4.3. Factors influencing observed trends

Possible reasons for the trends observed here may include: successful management strategies being employed at Gained sites associated with land use and land cover change (increased resilience); reduced pressures associated with land use and land cover change in Gained sites; legacy impacts; or factors associated with dataset scale and/or sampling error. Agri-environmental schemes and WFD programmes of measures contain measures that should lead to catchment resilience and improvements in water quality (Murphy et al., 2015). Richards et al. (2015) for example, reported reduced levels of nitrate leaching from farms subscribed to the Rural Environmental Protection Scheme (REPS) in Ireland, while Finn and O hUallacháin (2012) reported that nutrient management measures in REPS were expected to have a positive impact on water quality. Based on this, further investigations should assess the proportion of Lost and Gained sites that initiated improvement/management strategies beyond the data attributes in the CLC. However, as there is a lag time between measures being employed and any observed response from aquatic communities (Natural Scotland, 2014), this should be factored into any assessments made. Additionally, land use and land cover changes or activities that occurred prior to the assessment periods identified here, may be influential, and/or associated with legacy impacts. Withers et al. (2014), for example, reports how a legacy of accumulated unused P and N in the soil may provide for an omnipresent source of background nutrient loadings every time there is a large rainfall event, which may take decades to exhaust, even with a cessation of further fertilizer applications. Again, as previously mentioned, variation in management strategies, that are not detectable based solely on the use of land cover layers, requires further assessment.

When assessing the relationship between land cover and water quality, the scale at which the survey is carried out is an important factor (Buck et al., 2004). Sponseller et al. (2001) for example, found that the type of land cover present at a 200 m riparian scale, had a stronger impact on the invertebrate communities occurring, than land occurring beyond this zone, suggesting that changes at this near-stream level influence invertebrate community composition, regardless of land use and land cover changes outside of this zone. Additionally, Doody et al. (2012) and McDowell et al. (2014) highlight the importance of "critical source areas" (CSAs), as relatively minor sections within a field, farm or catchment that are responsible for the majority of diffuse pressures to water-bodies. In contrast, Sutherland et al. (2010) found that the percent of watershed (catchment scale) under agricultural land was a better indicator of the status of the riffle substrate condition than the percent of riparian land under agriculture. To rule out scale as a factor relating both Lost and Gained status to similar land cover change types, further studies should be carried out with finer land cover datasets. This should provide greater levels of detail than the 25 ha minimum mapping area of the CLC dataset. Furthermore, generating results by grouping all watersheds for each status type together (as in Tables 2.1-2.3, and Figures 2.4 and 2.5), may result in biasing, as the area of each individual watershed varies considerably. One or two larger watersheds may dictate the land use or land cover type that is occurring for the overall status classification. While proportional distribution is one way to counter act this, this was only employed in the creation of land cover change matrices, and prior to the use of all Mann-Whitney U and logistic regression techniques.

Limitations associated with sampling may also be a factor in detecting sites gaining or losing status. For example, the Scottish Environment Protection Agency (SEPA) and the UK Environment Agency (EA) monitor water-bodies in Spring and Autumn every year and then provide an average result over a number of years (Natural Scotland, 2014; WISER, 2015), whereas the EPA in Ireland only sample a water body once every three years (EPA, 2006)**.** In comparison to SEPA and the EA, the strategy employed by the EPA may result in larger fluctuations in the status of a waterbody, due to the extensive period between sampling, and again this should be further investigated as an explanation to the relationship between Lost and Gained land cover variables.

2.5. Conclusions

Developing an understanding of the relationship between land use/land cover trends and water-bodies is crucial for mitigating against potential stressors. While traditional water quality methods have included an evaluation of both the biological and chemical status, a more holistic approach is to include land use and land cover in the overall assessment. This study demonstrated potential methods to carry out such a land cover assessment. Here, land cover trends were linked to declines in water body status, through the overriding occurrence of anthropogenically influenced land (in comparison to the higher level of natural/semi-natural land occurring in Maintained sites). For example, in the period 2006-2012, Land that changed from Forestry to Heterogeneous Agricultural areas was 17.5 times more likely to result in Lost status, whereas land that remained as Forestry or remained as Inland Wetlands reduced the chance of Lost status occurring by 15 % and 4 %, respectively. However, the similarity of land cover trends between Lost and Gained status provides further research questions on: 1) possible measures being implemented in catchments with Gained status; and 2) the efficacy of the ecological sample survey resolution to adequately detect trajectories of change. Based on this, the need for future studies to assess the influence of management strategies, land use intensity, scale and sampling error/frequency were highlighted.

Chapter 3

3. Investigating hydrological pressures on high status rivers

3.1. Introduction

The hydrological pattern of a river system has been described as a “master variable” that is responsible for driving many physical and biological characteristics within the river (Poff et al., 1997; Richter et al., 2003; Poff and Zimmerman, 2010). These physical and biological characteristics include nutrient cycling (Richter et al., 2003; Jones et al., 2015), temperature changes (Webb et al., 2008), rates of sedimentation (Growns et al., 2017), habitat modification and subsequent species diversity (Extence et al., 1999; Carlisle et al., 2011; Mims and Olden, 2013). However, over the past century, human activities, either directly in the form of land use change (Schilling et al., 2008; Schottler et al., 2014) or indirectly through climate change and changes in precipitation levels (Barnett et al., 2008), have profoundly affected hydrological regimes (Wang and Hejazi, 2011). Vörösmarty et al. (2010) for example, reported that 65% of global discharge and associated habitats are under threat, and these changes are predicted to continue in the future (e.g. Döll and Schmied, 2012; Laizé et al., 2014).

In the United States of America (USA), an assessment of 2888 streams by Carlisle et al. (2011) found 86% to have altered streamflow magnitudes in comparison to reference sites, while across the European Union (EU), hydromorphological adaptions and related habitat changes, are key pressures associated with over 40% of Water Framework Directive (WFD – OJEC, 2000) classified rivers, 40% of transitional waters, and 33% of lakes (EEA, 2012). These hydromorphological adaptions are driven by urban development, flood control, hydropower generation, channelization, and land drainage (EEA, 2012). Several studies have reported a significant negative relationship between invertebrate community compositions and the extent of streamflow alteration (e.g. Carlisle et al., 2014; Growns et al., 2017). Furthermore, a world-wide assessment of the effect of dams on large river systems (Nilsson et al., 2005) highlighted catchment-scale impacts such as the destruction of ecosystems and obstructions to fish migration, as well as increased nutrient and sedimentation discharges. Poff and Zimmermann (2010), in a review of the literature (165 papers) assessing ecological responses to flow modification, found 92% reported negative ecological impacts. A similar review by Bunn and Arthington (2002) lists: modification to habitat and subsequent impacts on biotic species composition; disruptions to species life cycle strategies; changes to / loss of connectivity between aquatic habitats; and an easier proliferation of habitats by non-native invasive species; as four key principles associated with altered flow impacts on aquatic biodiversity in river and stream habitats.

Flow alterations related to land use change are especially associated with agricultural practices, forestry and urbanisation (Prévost et al., 1999; Malmqvist and Rundle, 2002; Allan, 2004). These alterations may occur through removal of vegetated cover, which has implications for the runoff-evapotranspiration balance (Schilling et al., 2008), as well as soil erosion. Drainage, through channelization, ditching and the introduction of subsurface networks, may also result in changes to seasonal runoff patterns including high flows and base-flows (Prévost et al., 1999; Blann et al., 2009), while the construction of roads, roofs, and carparks increases the impermeable surface areas of catchments (Allan, 2004). For example, less evapotranspiration loss occurs from seasonal crops compared to land with continuous vegetation cover, which ultimately results in an altered water balance and increased streamflow (Schilling et al., 2008). Drainage of land for agricultural purposes, especially that of peat soils, enables cultivation and increases agronomic carrying capacity and yield potential (Paul et al., 2018). However, in comparison to a natural undrained watershed, the introduction of drainage reduces the residence time of water in the soil (Schottler et al., 2014), so for example, precipitation in winter is removed more rapidly leading to increased winter streamflow rates and a reduction in summer streamflows due to a lack of stored water (Blann et al., 2009). This may potentially lead to more erosive rivers (Schottler et al., 2014) and the problem is enhanced given for example, that water draining agricultural land is potentially the carrier of nutrient pollutants (Schilling et al., 2008; Collins et al., 2018) along with sediment and organic carbon (Glendell and Brazier, 2014).

While drainage is one factor, precipitation and climate change have also been cited as drivers of changes to streamflow (Scheurer et al., 2008; Wang and Hejazi, 2011; Schottler et al., 2014). Barnett et al. (2008) for example, demonstrated that increased spring flows and decreased summer flows in the period 1950 to 1999 in the Western USA region, was 60 % associated with human induced climate change. In Europe, increasing streamflow trends in Northern Regions and decreasing trends in the South and East during the period 1962 to 2004, which coincide with similar patterns in precipitation, were demonstrated by Stahl et al. (2010), although potential cofounding factors such as pressures associated with land use and land cover change should also be taken into account. Separating the impacts of climate change and changes in precipitation rates from those of land use and land cover change is difficult, as in many cases each counter-acts the other (Jaramillo and Destouni, 2014). However, Wang and

Hejazi (2011) highlighted how some studies focusing on climate change or precipitation tend to exclude other human influences, such as abstractions.

High status water-bodies (HSWs) as designated under the EU WFD (OJEC, 2000) are rivers, lakes, transitional and coastal waters, and represent conditions that are close to natural, having suffered little/no anthropogenic impacts (WG 2.3, 2003; Mayes and Codling, 2009). However, as with all water-bodies, HSWs are under threat from nutrient enrichment, sedimentation, priority substances, flow alterations, habitat loss, invasive species, and unsustainable use (Malmqvist and Rundle, 2002; Dudgeon et al., 2006; Poff and Zimmerman, 2010; Vörösmarty et al., 2010; Collen et al., 2014). These stressors are very often from adjacent land use practices (Allan, 2004; Foley et al., 2005; Poole et al., 2013; Lange et al., 2014). Ireland, which has a relatively high proportion of HSW sites, in comparison to other EU countries (based on data extracted from the Europe (WISE) - WFD database - EEA, 2015), has seen large deteriorations in recent years. While some reversals in deterioration have been observed, the trend is an increasingly negative one (EPA, 2012; EPA, 2016). For example, between 1998 to 2009, 358 river sites had deteriorated from high status, with counties Donegal (-79 sites), Mayo (-33 sites) and Sligo (-31 sites) being particularly badly impacted (White et al., 2014). Ní Chatháin et al. (2012) and White et al. (2014) highlighted that, in contrast to already degraded waterbodies, minimal increases in pressure from, for example, nutrient enrichment, sedimentation or streamflow modification, are likely to have a disproportionately large impact on HSWs, and it is this potential relationship between HSW deteriorations and streamflow modifications that was examined here.

The use of macro-invertebrates for assessing water quality is a widely accepted practice, and they have been employed for assessing: nutrient enrichment (Guilpart et al., 2012); sedimentation (Extence et al., 2013; Glendell and Brazier, 2014; Murphy et al., 2015); acid mine drainage (Van Damme et al., 2008); as well as streamflow alterations (Carlisle et al., 2014; Growns et al., 2017). Gieswein et al. (2017) for example, found macro-invertebrates to be more responsive than macrophytes or fish in an assessment of multiple anthropogenic stressors, although Carlisle et al. (2011) found both fish and invertebrates useful for assessing alterations in streamflow. Several countries have developed biotic metrics that employ invertebrates specifically to assess how hydrological/flow changes have impacted on riverine ecology, e.g. Canada (Armanini et al., 2011), Estonia (Timm et al., 2011), New Zealand (Greenwood et al., 2016), and for temporary streams in the Mediterranean Basin (Cid et al., 2016). Several of these metrics have drawn on the Lotic Index for Flow Evaluation – LIFE metric as developed by Extence et al. (1999).

With this background, and with regard to the decreasing condition of HSWs in Ireland as a case study, the aim of this study was to investigate the evidence of hydrological (streamflow) change as a pressure on river biology in Irish HSW rivers. The objectives were to: 1) use invertebrates and the LIFE index as developed by Extence et al. (1999), to assess if streamflow alteration has impacted on river biology; 2) assess the historical relationship between streamflow data and the change in status of HSW rivers; and 3) assess the historical relationship between extant precipitation data and the change in status of HSW rivers.

3.2. Materials and Methods

3.2.1. Sampling

Macro-invertebrate sampling was carried out at sixty-five high status river sites selected at random (although coded for typology - slope and hardness) from 167 West of Ireland high status river sites (Figure 3.1 and Appendix B). Based on Environmental Protection Agency (EPA) ecological quality (Q-value) analysis (primarily utilising macro-invertebrate communities), these sixty-five sites were determined (following the example of Roberts et al., 2016) to have either: Lost their high status (e.g. gone from high to good, moderate, poor or bad); consistently Maintained high status; or had Gained in status (e.g. from good to high) – see Appendix B for additional information. This resulted in the sampling of 20 Lost sites, 24 Maintained sites and 21 Gained sites.

Sampling was carried out in April/May (Spring) 2016 and in August (Summer) 2016 and this was repeated in 2017, thereby allowing for seasonal and yearly trends to be determined. Macro-invertebrates were collected by three-minute kick-sampling, followed by a one-minute stone searching, using methods described in BS-ISO (2012) and Environment Agency (2012). As a reference point, on each sampling occasion, discharge monitoring based on the velocity-area method, was carried out using an OTT MF Pro flow meter. However, due to dangerous sampling conditions, three sites were excluded from sampling in Summer 2016 and in Summer 2017.

3.2.2. Streamflow assessment - LIFE scores

Macro-invertebrates were identified to species level, with the exception of Oligochaetes (Order) and Dipterans (Family or Tribe), using an Olympus (SZX16) stereo microscope and relevant dichotomous keys (see Appendix G). Following identification, the Lotic-invertebrate Index for Flow Evaluation (LIFE) method as described by Extence et al. (1999) was applied. This method, which is based on the recognised relationships between differing flow types and their corresponding macro-invertebrate communities, assigns macro-invertebrate taxa (either at species or family level) to flow groups one to six depending on their associated flow preferences. Monk et al. (2012) highlights the benefits of using species over families when applying the LIFE method, with for example, taxa within families having different flow type preferences. The flow groups one to six represent rapid, moderate/fast, slow/sluggish, flowing/standing, standing, or drought resistant, taxa preferences respectively. Based on these groups and the abundance at which taxa occur, a LIFE score is generated. Taxa associated with rapid/faster flows score higher, while standing water/drought resistant taxa score lower. The metric is calculated as:

$$LIFE = \frac{\sum fs}{n} \qquad \text{[Eq. 3.1]}$$

where $\sum fs$ is the sum of the flow scores determined from a list of taxa and scores in the Appendix of Extence et al. (1999), and n is the number of scoring families. Typically, higher flows are related to higher LIFE scores (Extence et al., 1999). In the original study, Extence et al. (1999) define ‘flow’ as velocity, although within the context of this study, and from hereafter flow, is interpreted as streamflow or ‘discharge’ (see also Dunbar et al., 2010).

Using the R software programme (R Core Team, 2018), Wilcoxon-Mann-Whitney tests, were used to test for differences between the LIFE scores of different status categories, e.g. Lost against Maintained, within each sample period, as the data were not normally distributed and were non-transformable. Wilcoxon-Mann-Whitney tests were also used to test for direction of differences between categories (i.e. greater or less than). Wilcox Signed Rank tests were used to test for differences in LIFE scores between seasons and also between years, and this was also repeated within each status category, and again the direction of change was also analysed. Some caution is required with the interpretation of these results, as both the classification of status categories and the generation of LIFE scores employ invertebrates, and so are not therefore, fully independent of each other. However, while the generation of status categories, through the EPA Q-value system, is more aimed at assessing general/organic pollution patterns, the generation of LIFE scores is specifically related to the preference of invertebrate taxa for specific flow types.

3.2.3. Generation of historical LIFE scores

Using EPA monitoring data for the sampling periods 2007, 2008 and 2009 (labelled 2009A) and for the sampling periods 2010, 2011 and 2012 (labelled 2012A), historical LIFE scores were generated for 286 of the high status river sites recorded through-out Ireland (Figure 3.2). EPA monitoring of macro-invertebrates using the Q-value system identifies macro-invertebrates as occurring as either Present (1-2 individuals), Scarce/Few (<1% of the total sample), Small numbers (<5%), Fair numbers (5-10%), Common (10-20%), Numerous (25-50%), Dominant (50-75%) and Excessive (>75%) (McGarrigle et al., 2002). However, within the dataset for the 286 sites, only the abundance categories “single, few, common, numerous and dominant” were present. Unmodified, these EPA abundance categories are incompatible with the LIFE metric, which categorises invertebrate abundances within a log scale, i.e. of 1-9, 10-99, 100-

999, 1000-9999, and <10,000 (Extence et al., 1999). Therefore, in order to generate LIFE scores from the EPA datasets (and incorporating as a harmonising assumption), the EPA abundance categories were re-assigned as: Single and Few = 1-9; Common = 10-99; Numerous = 100-999; and Dominant = >999.

As a check on this assumption, actual LIFE score data, generated from the Summer 2016 sampling period, were compared with a LIFE score that was generated using the same invertebrates, that were given as a percentage of the total invertebrate count for that sample. The percentage abundance was then assigned as either Single, Few, Common etc., depending on the percentage category. The percentage categories were: 0 > Few < 9% ; >=10 % Common =< 22% ; >= 23% Numerous <= 50% ; > 50% Dominant <= 75% ; and > 75% = Excessive. These categories were then assigned as, for example, Few = 1-9, Common = 10-99, etc. A Spearman rank correlation of the actual LIFE scores against the generated LIFE scores had a correlation coefficient of 0.849 at $p<0.01$, indicating a very strong relationship. A stronger relationship (0.999 at $p<0.01$) was observed when the generated LIFE score was created by directly assigning the categories Few, Common, etc. based on the numbers of taxa present, without first calculating the abundance percentages. Plots of the correlations of actual LIFE scores from Summer 2016 against the generated LIFE scores from Summer 2016 are presented in Appendix C Figure C21.

Historical LIFE scores for the periods 2009A and 2012A, for the 286 high status sites were compared with each other, with paired Wilcox Signed Rank tests used to analyse the data.

3.2.4. Streamflow – hydrometric stations

Using GIS, the location of the nearest hydrometric/flow monitoring station with available long-term (>= 10 years) flow monitoring data, to fifty-nine of the sixty-five invertebrate sampling sites was determined (Figure 3.1) (i.e. six sites did not have any stations). In many instances, due in part to the remoteness of many of the sample sites, the hydrometric stations were either not located on the same waterbody, or were positioned at a considerable distance from the sample sites. However, they were employed to generate general streamflow trends for the associated sample sites based on the assumption that general hydrological patterns may have shown regional similarities across landscapes. Appendix C provides relevant locational information for the twenty hydrometric stations employed (i.e. several of the fifty-nine sample sites shared hydrometric stations). Daily discharge data for these hydrometric stations were obtained from the EPA Hydronet website[4], which also encompasses/links to other flow monitoring authorities, such as the Office of Public Works (OPW), the Marine Institute, and the Electrical Supply Board (ESB). The R package "Flowscreen", which is designed to summarise daily streamflow time series data and identify significant trends (Dierauer and Whitfield, 2017), was used to generate and identify trends in base-flow variables such as annual mean baseflow, annual baseflow volume, annual maximum baseflow, annual minimum baseflow, and mean baseflow index between the period 1998-2018. These trends (grouped as either increasing, decreasing or no trend) were analysed relative to the status (Gained, Lost, or Maintained) of the waterbody, using a Chi-squared text. At the seasonal level, the Q5, Q10, Q95 and Q5/Q95 discharge percentiles for the daily discharge data were determined for the winter months December, January, February and March (DJFM) and summer months

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[4] http://www.epa.ie/hydronet/#Water%20Levels

June, July, August and September (JJAS) over the period 1998-2018. A Kruskal–Wallis test was used to test for differences between these percentiles and the three status categories. Additionally, annual trends for each individual percentile were analysed over the period 1998-2018 using the Mann-Kendall Trend test for assessing trends in time-series data, via the R package "Kendall" (McLeod, 2015). Durbin-Watson tests via the R package "lmtest" (Hothorn et al., 2019) were conducted on the datasets used for the Mann-Kendall test to ensure that the assumption of no serial correlation in the datasets test was met. Where serial correlation was present, a Modified Mann-Kendall Test for Serially Correlated Data (MMKH) was conducted, via the R package "modifiedmk" (Patakamuri and O' Brien, 2019).

3.2.5. Rainfall data

Using GIS, the location of the nearest rainfall monitoring station to the hydrometric station was determined (Figure 3.1 and Appendix C). Rainfall data for these monitoring stations were downloaded from the Met Éireann website[5]. The Mann-Kendall Trend test for assessing seasonal trends in time-series data, via the R package "Kendall", was used to assess annual rainfall trends and winter DJFM and summer JJAS trends over the 1998-2018 period. Again, Durbin-Watson tests via the R package "lmtest" (Hothorn et al., 2019) were conducted on the datasets used for the Mann-Kendall test to ensure that the assumption of no serial correlation in the datasets test was met. Where serial correlation was present, a "Modified Mann-Kendall Test for Serially Correlated Data (MMKH)" was conducted, via the R package "modifiedmk" (Patakamuri and O' Brien, 2019).

[5] https://www.met.ie/climate/available-data/historical-data

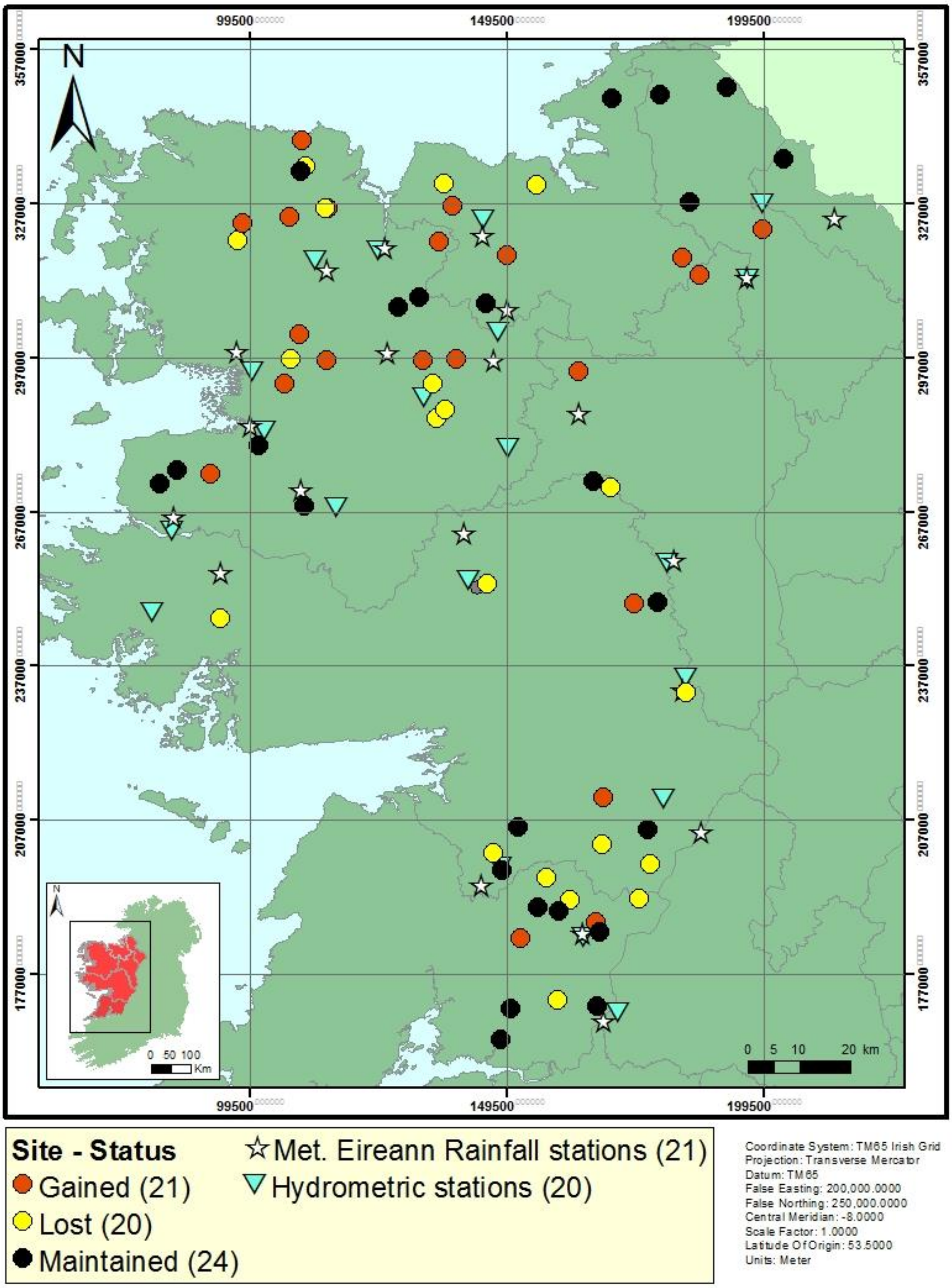


Figure 3.1. Location of the sixty-five sampling sites and the nearest hydrometric station and rainfall monitoring stations.

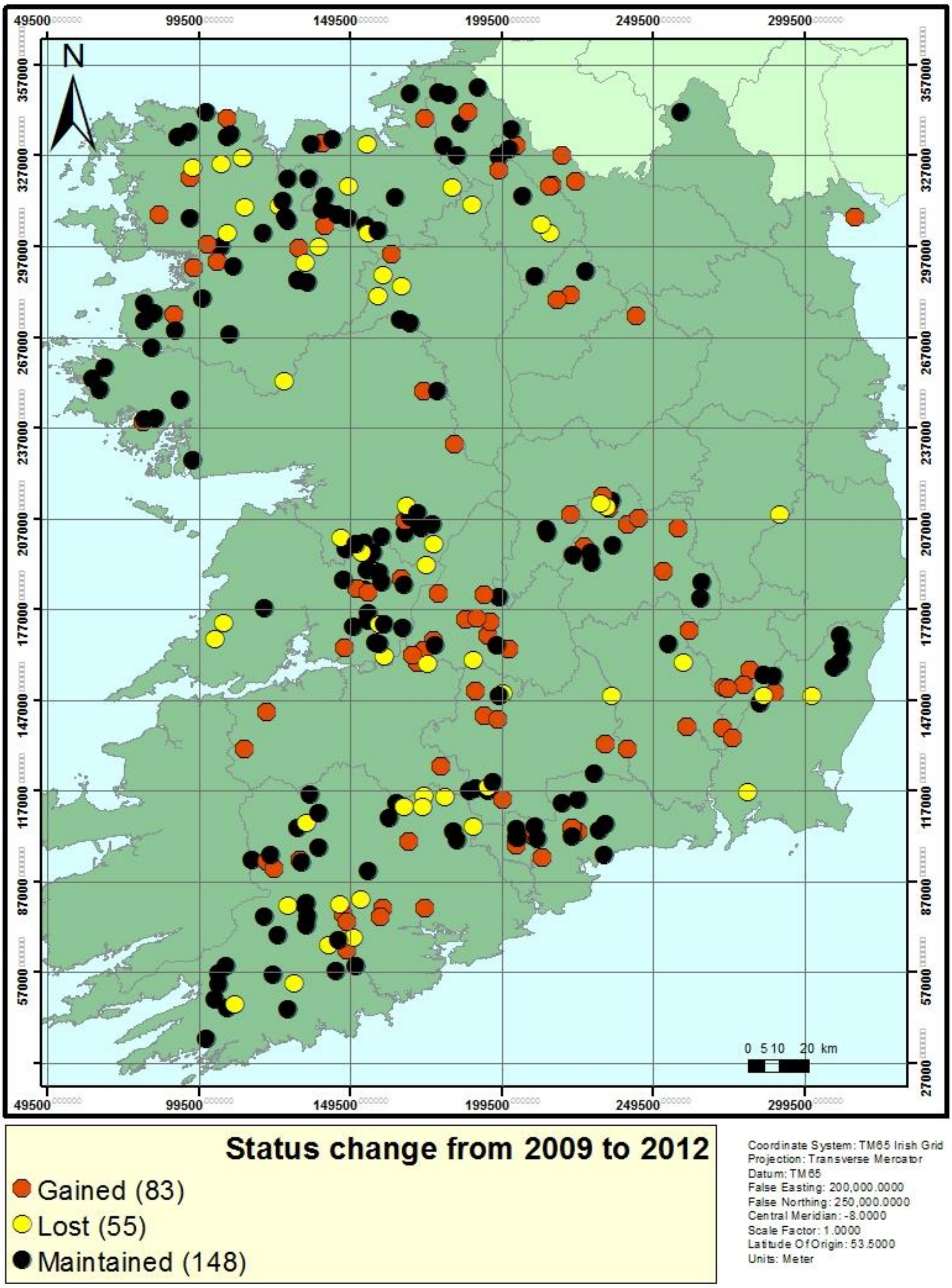


Figure 3.2. Location of the 286 sites for which historical LIFE scores were generated.

3.3. Results

3.3.1. Invertebrate communities present

A full inventory of all macro-invertebrates found during sampling is provided in Appendix E. The average number of individual taxa types occurring in each sample based on their preference for flow types, and the overall abundances of taxa occurring based on their preference for flow types, are presented in Figures 3.3 and 3.4, respectively.

In Spring 2016, for each sample site, the majority of individual invertebrate taxa types present were taxa associated with moderate/fast flowing waters (group two from Extence et al., 1999). Only one site (35F010100 – Lost) had a greater number of taxa associated with a rapid flow type (group one). Based on abundances, the majority of samples again had higher abundances of group two taxa, although seven Maintained, one Lost and one Gained sites had higher abundances of group one taxa, while two Lost sites had a higher abundance of taxa with a preference for slow/sluggish (group three) flowing waters. Elmidae, primarily *Elmis aenea* and *Limnius volckmari* (both group two), *Baetis rhodani* (group two), and *Rhithrogena* sp. (group one) were the most abundant taxa in twenty-six, eighteen and twelve sites respectively. Three sites had *Gammarus dubini* (group three) as the most abundant taxa, while Simuliidae (group two) were the most abundant at two sites. One Lost site had *Siphonoperla (Chloroperla) torrentium* (group one) as the most abundant taxa. The remaining sites had either, Chironomidae or Oligochieta as the most abundant taxa but these taxa are not scored in the LIFE metric.

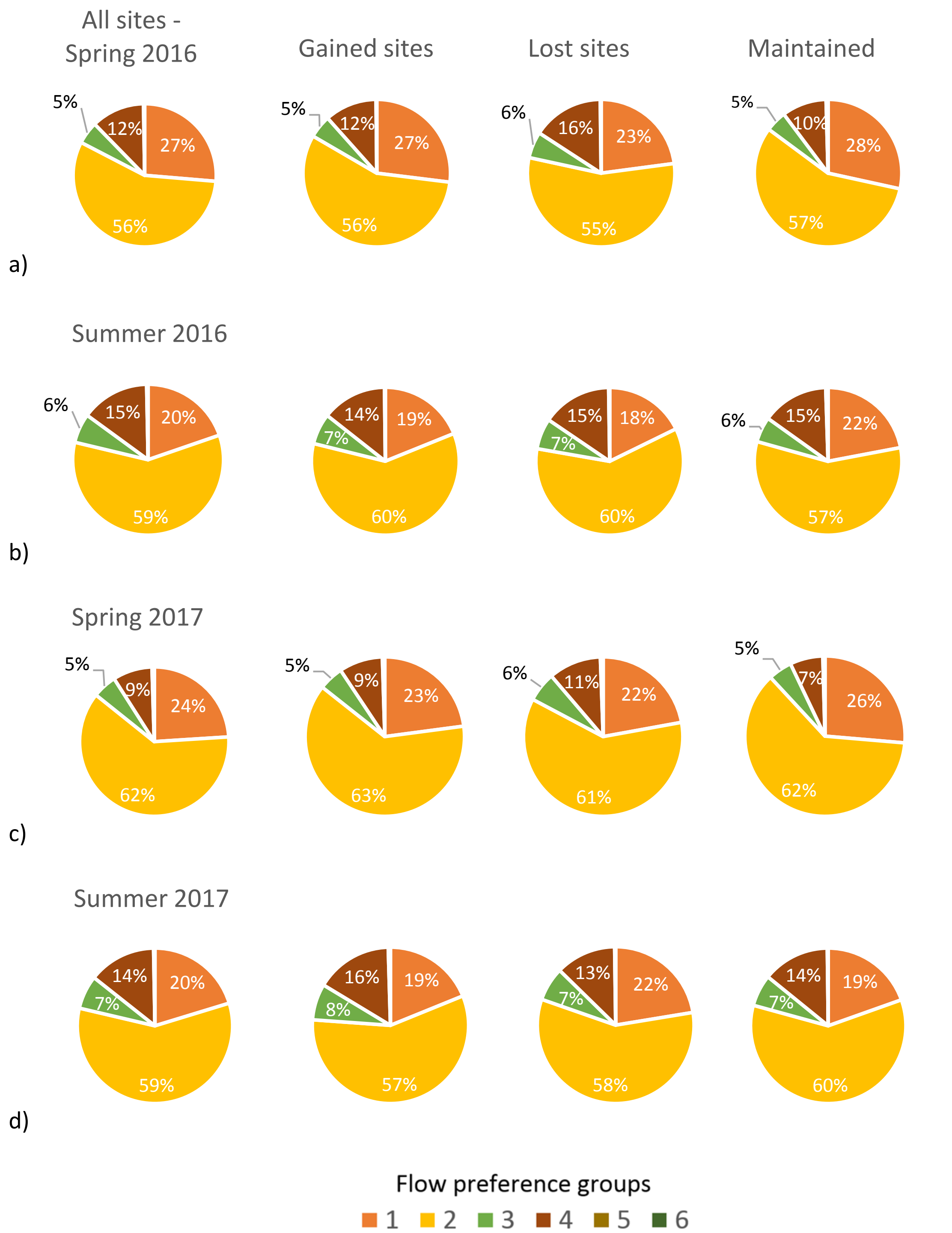


Figure 3.3. Average number of individual taxa types occurring based on their preference for flow types 1) Rapid; 2) Moderate/Fast; 3) Slow/sluggish; 4) Flowing/standing; 5) Standing; and 6) Drought resistant; in samples from a) Spring 2016, b) Summer 2016, c) Spring 2017 and d) Summer 2017.

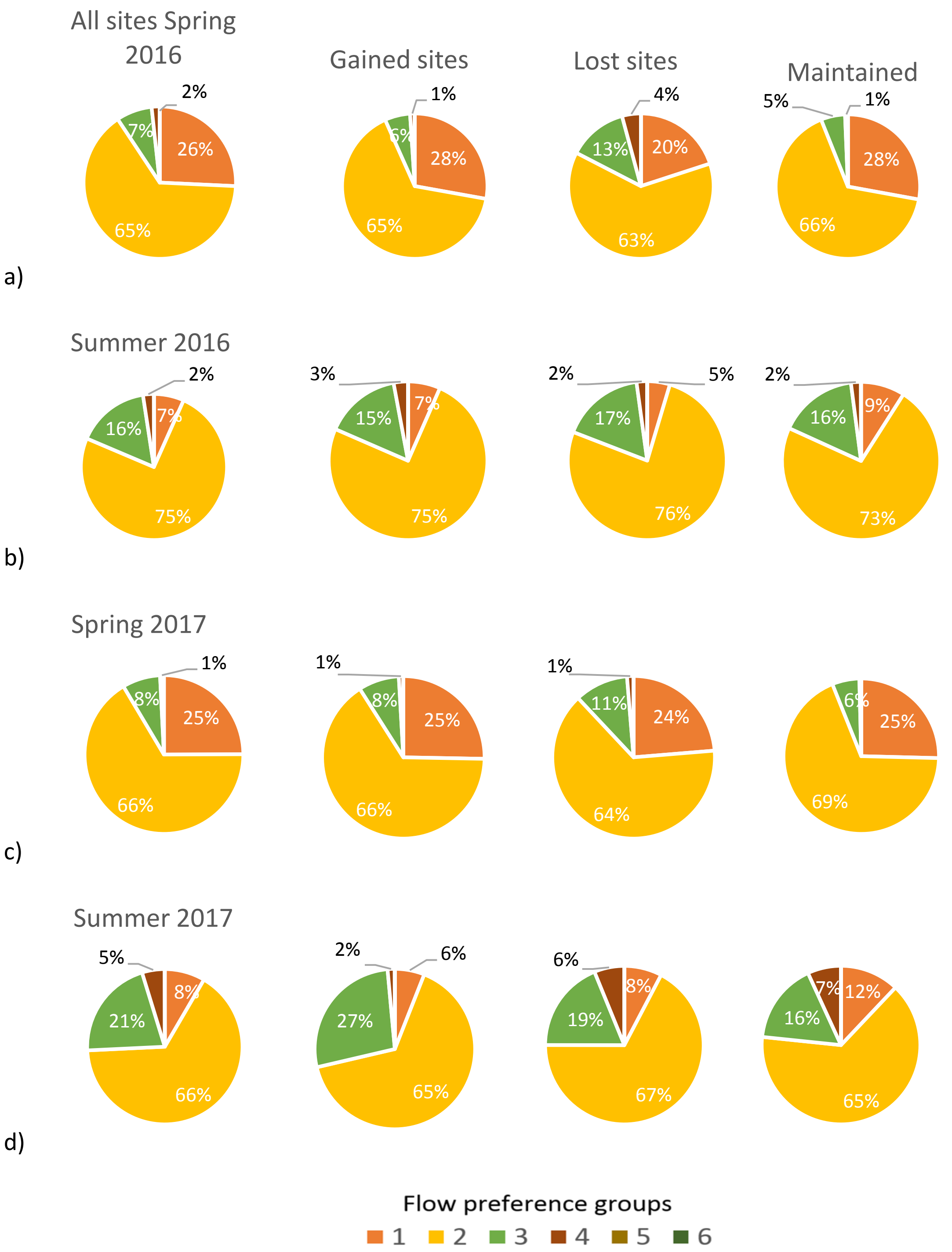


Figure 3.4. Abundances of taxa occurring in the sixty-five samples sites based on their preference for flow types where: 1) Rapid; 2) Moderate/Fast; 3) Slow/sluggish; 4) Flowing/standing; 5) Standing; and 6) Drought resistant; in samples from a) Spring 2016, b) Summer 2016, c) Spring 2017 and d) Summer 2017.

Similarly, in Summer 2016, the majority of individual invertebrate taxa types present at each site were taxa associated with moderate/fast flowing waters (group two). Again, abundances of group two taxa were higher than any other group, with the exception of one Maintained site, which had higher abundances of group one taxa, and two sites (Maintained and Gained) that had higher abundances of group three taxa. Elmidae, again primarily *Elmis aenea* and *Limnius volckmari*, and *Baetis rhodani* were the most abundant taxa, occurring in forty-one and six sites, respectively. *Gammarus dubini*, *Seratella ignita* (group two), *Leuctra fusca* (group two), *Agapetus* sp. (group two) and the snail *Potamopyrgus antipodarum* (flowing/standing waters - group four) were the scoring taxa that were most abundant in the remaining sites. Group two taxa again dominated samples collected in Spring 2017 and Summer 2017 in terms of species types present and abundances although the abundances of group one taxa were higher in eight of the sixty-five Spring 2017 sample sites. In Summer 2017, only two sites had taxa of greater abundances from a group other than group two.

3.3.2. Recent LIFE scores (2016 and 2017)

The average LIFE scores for each status category for each sample period, and the average number of scoring taxa for each status category are presented in Table 3.1 (see also Appendix C for full list of LIFE scores). All sixty-five sample sites, across all sampling periods, had a LIFE score above 7.25, with only six sites in Spring 2016, twelve sites in Summer 2016, one site in Spring 2017 and eleven sites in Summer 2017, having scores below eight. The lowest life score, 7.31, was found in a Maintained site in Summer 2016, while the highest score, 9.04, was similarly found in a Maintained site in Spring 2016.

Table 3.1. The average LIFE scores for each status category for each sample period, and the average number of scoring taxa (n) for each status category, with standard deviation in parenthesis.

	Spring 2016		**Summer 2016**		**Spring 2017**		**Summer 2017**	
	LIFE	**n**	**LIFE**	**n**	**LIFE**	**n**	**LIFE**	**n**
All	8.3 (0.3)	27 (5.3)	8.2 (0.3)	25 (6.3)	8.4 (0.2)	28 (6.0)	8.2 (0.3)	22 (5.9)
Gained	8.4 (0.2)	28 (4.6)	8.2 (0.2)	27 (7.2)	8.4 (0.2)	29 (5.3)	8.2 (0.2)	23 (7.3)
Lost	8.2 (0.3)	26 (5.5)	8.2 (0.3)	24 (6.0)	8.4 (0.2)	26 (7.4)	8.0 (0.2)	21 (5.3)
Maintained	8.4 (0.3)	28 (5.3)	8.2 (0.3)	25 (5.3)	8.4 (0.2)	28 (4.8)	8.3 (0.2)	22 (4.7)

A significant difference in LIFE scores between Maintained and Lost sites was observed in Spring 2016 ($p = 0.038$), Spring 2017 ($p = 0.012$), and Summer 2017 ($p < 0.01$). On all occasions Maintained sites had greater LIFE scores with p values of 0.019, $p<0.01$, and $p<0.01$, respectively. No difference was found in Spring 2016 or Spring 2017 between Lost and Gained sites, and Maintained and Gained sites. In Summer 2016 no significant difference in LIFE scores between any of the status categories was found. In Summer 2017 a significant difference between Lost and Gained was observed ($p = 0.022$), with Lost scores being less than Gained scores ($p = 0.011$). No difference was found between Gained and Maintained.

Analysis over the two years of sampling found a significant difference between Life scores from Spring 2016 and Spring 2017 (Wilcoxon signed rank paired test, $p = 0.018$), but not between Summer 2016 and Summer 2017. Spring 2016 scores were less than those of Spring 2017 ($p < 0.01$). Seasonal analysis found a significant difference between Spring 2016 and Summer 2016 and between Spring 2017 and Summer 2017 (both $p < 0.01$), with Spring scores being greater than Summer scores for both years ($p < 0.01$). Figure 3.3 shows only a slight change in the distribution of individual taxa types occurring between seasons based on their flow group preferences, but there is a clear seasonal change in abundances of taxa occurring (Figure 3.4).

Within the Maintained category there was no significant difference between Maintained LIFE scores in Spring 2016 and Spring 2017, and Summer 2016 and Summer 2017, but there were seasonal differences for both years (Wilcoxon signed rank paired test, $p < 0.01$). Within the Lost category, yearly differences occurred between the two Summer samples ($p < 0.01$), but not the Spring samples, while seasonal differences were found between Spring and Summer 2017 ($p < 0.01$), but not in 2016. Within the Gained category, yearly differences occurred between the two Spring samples ($p = 0.025$), but not the Summer samples. Seasonal differences were found in Gained sites between Spring and Summer for both 2016 ($p=0.014$) and 2017 ($p<0.01$). For all significant results, Spring 2016 scores were less than those of Spring 2017, while Summer 2016 scores were greater than Summer 2017. Again, for both years, Spring scores were greater than Summer scores.

3.3.3. Historical LIFE scores

For LIFE scores generated for the sampling periods 2007, 2008 and 2009 (labelled 2009A) and for the sampling periods 2010, 2011 and 2012 (labelled 2012A) from EPA monitoring data for 286 of the high status sites recorded through-out Ireland, only one site had a score below 7.25. For the 2009A period, site 36D070100 had a LIFE score of 6.86, although this increased to 8.5 for the 2012A period. The average score across all 286 sites in the 2009A period was 8.32 with a minimum score of 6.86 and a maximum of 9.33. In 2012A the average score was 8.37 with a minimum score of 7.25 and a maximum again of 9.33. A significant difference between all of the 2009A and 2012A scores was found (Wilcoxon paired test, $p<0.01$), with the scores in 2009A being less than 2012A ($p<0.01$). Of the 286 sites, 148 sites continuously maintained a high status rating (e.g. Q-value of 4.5 or 5) over the 2009A and 2012A periods, for which no significant difference between the LIFE scores of 2009A and 2012A were found (Wilcoxon paired test). Eighty-three sites improved from below high to high status (e.g. from Q-value 3, 3.5 or 4 to 4.5) and within these sites a significant difference in LIFE scores between 2009A and 2012A was found (Wilcoxon paired test, $p<0.01$), with 2009A scores being less than 2012A ($p < 0.01$). Fifty-five sites deteriorated from high to below high status between 2009A to 2012A but no significant difference in LIFE scores between 2009A and 2012A for these sites was found.

3.3.4. Rainfall trends

In the 20 year period from 01/01/1998 to 01/01/2018 a changing trend in rainfall pattern (either increasing or decreasing) was found from only one rainfall measuring station associated with the fifty-nine sample sites/hydrometric stations, 6819 (tau = 0.45, $p < 0.01$, Mann-Kendall trend test). In the same 20 year period for both the JJAS and DJFM time periods, no significant changing trend in rainfall patterns (either increasing or decreasing) was found, although monitoring station 6819 had increased rainfall trends in JJAS and DJFM with p-values of 0.059 and 0.064 respectively, while station 2218 had an increasing rainfall trend in DJFM with a p-value of 0.053.

3.3.5. Streamflow assessment - hydrometric stations

Of the hydrometric stations associated with the fifty-nine sample, two stations, 25030 and 29071, had an increase in annual mean daily discharge, although this was at p values of 0.09 and 0.06, respectively sites (see Appendix C for “Flowscreen” outputs for each hydrometric station and for location details). More significantly, these two stations also had an increase in annual mean baseflow, at $p = 0.01$ and $p = 0.03$, respectively. Three stations, 30007, 30020 and 31072, had an increase in annual baseflow volume, although only station 30020 ($p = 0.02$) was significant at the level of p =< 0.05. Both 30007 and 31072 had increases at p values of 0.10. The same three stations (with the same p values) had increases in annual maximum baseflow. Five stations, 26002 ($p = 0.05$), 26029, 26030 (p =< 0.01), 32026 ($p = 0.01$) and 34007 (p =< 0.01) had a decrease in annual minimum baseflow, although for station 26029 this was at a significance of $p = 0.10$. Station 25020 ($p = 0.03$) had an increase in mean baseflow index. Chi-squared tests between status and each of the base-flow variables (grouped as either increasing; decreasing - annual minimum baseflow only; or no trend), revealed no statistical difference between the different status categories and the baseflow characteristics.

Using a Kruskal –Wallis test, no significant difference was found between the three status categories and the percentiles Q5, Q10, Q90, Q95 or Q5/Q95 for both the periods JJAS and DJFM, i.e. there was no difference in the median value of Q5/Q95 for all three status categories. Additionally, for the aforementioned percentiles for the periods Summer (JJAS) and Winter (DJFM), no significant difference was found between Gained and Maintained, Maintained and Lost and Lost and Gained (Wilcoxon-test). During the 20 year period, from 1998 up to 2018, in JJAS, five sites displayed a significant changing trend in percentiles, with three sites showing an increase in Q90 and Q95 similar to the prior baseflow analyses, and two sites showing a decrease (Table 3.2). Nine sites, in DJFM, displayed a significant changing trend in discharge percentiles (Table 3.3).

Table 3.2. Trends in percentiles during the period JJAS over 20 years from 1998-2018 for the hydrometric stations associated with the sixty-five (sixty) sample sites.

Site	Percentile	Tau	P-value
26010	95	0.467	0.018
26010	90	0.410	0.038
26010	Q5/Q95	-0.562	0.004
27001	95	0.358	0.030
27001	90	0.358	0.030
26007	95	0.367	0.053
26007	90	0.383	0.043
26030	95	-0.618	0.001
26030	90	-0.544	0.003
34007	95	-0.442	0.007
34007	90	-0.453	0.006
34007	Q5/Q95	0.358	0.030

Table 3.3. Trends in percentiles during the period DJFM over 20 years from 1998-2018 for the hydrometric stations associated with the sixty-five (sixty) sample sites.

Site	Percentile	Tau	P-value
6030	Q95	0.347	0.042
15021	Q10	0.328	0.054
15021	Q5/Q95	0.509	0.003
25030	Q99	0.238	0.029
26010	Q95	0.486	0.013
26010	Q90	0.524	0.008
26010	Q5/Q95	-0.467	0.018
27001	Q95	0.358	0.030
27001	Q90	0.358	0.030
26007	Q5/Q95	0.383	0.043
30007	Q95	-0.474	0.004
30007	Q90	-0.389	0.018
30007	Q5/Q95	0.432	0.009
30020	Q95	-0.379	0.021
30020	Q90	-0.316	0.056
30020	Q5/Q95	0.379	0.021
34024	Q95	-0.324	0.059

3.4. Discussion

The general trend across all sampling periods was for group two taxa, which are taxa associated with moderate/fast flowing waters, to dominate in terms of taxa present and abundances. This was also reflected in the LIFE scores, which were all above 7.25, and again indicative of rivers associated with moderate/fast flowing waters. Similarly, with the exception of one site, the historical LIFE scores from the EPA derived data-set of 286 sites were also 7.25 or higher. In line with this, taxa such as *Elmis anena*, *Limnius volckmari* from the family Elmidae, which have a preference for riffle sections (Nilsson, 2005), *Baetis rhodani*, along with other group two taxa tended to have high abundances through-out. As these are/were HSW rivers, pollution sensitive taxa from the family Heptageniidae such *Heptagenia sulphurea*, *Ecdyonorus* sp. and *Rhithrogena* sp. along with Stoneflies *Perla bipunctata*, *Isoperla grammatica*, and the Chloroperlidae species *Siphonoperla torrentium* occurred in relatively high numbers in many sites, with each of these species belonging to the LIFE flow group one. Dunbar et al. (2010) describes how taxa from LIFE flow groups one and two have a tendency to dominate in unmodified and heterogenic river habitats, likely because of their relatively narrow niche habit requirements, e.g. fast-flowing high oxygenated water, while in contrast, taxa with a preference for slower flowing waters (flow groups three to six) occur more in modified water-bodies. Given that the study sites employed here, are or were high status, most, based on field work inspections, were unmodified with close to natural morphology, and it is likely that the un-observed sites from the EPA 286 data-set follow a similar description. It should be noted, however, that Growns et al. (2017) found several invertebrate taxa occurring in streamflow mesocosms contrary to those described by Extence et al. (1999) and suggest that the flow preferences of many taxa may be more flexible than previously assumed. This may in part explain some discrepancies in results discussed below for example, Lost versus Gained.

Here, significant differences in LIFE scores between sample sites that Maintained and Lost high status were observed in three sampling periods, with Maintained sites having greater LIFE scores on each occasion. Higher LIFE scores are associated with higher flows (Extence et al., 1999) implying that Maintained sites had higher streamflow rates than Lost sites. White et al. (2014) describes how a small change in pressure may have a big impact on HSWs. However, despite the differences between Lost and Maintained, all scores were generally in the same range, i.e. a minimum of 7.31 and a maximum of 9. An additional important caveat is the lack of any statistical difference between sites that had Gained in status and Lost status for three of the sampling periods, and the close relationship between the LIFE scores for Lost and Maintained from the Summer 2016 sampling period (i.e. p value of 0.8).

From the EPA dataset, of the fifty-five sites that Lost high status between 2009A to 2012A, no significant difference in LIFE scores between that period was found. This implies that streamflow alterations, or at least the hydrological impacts on invertebrates, was not a factor in the deterioration of these sites. In contrast, sites that Gained during this period, did show a significant difference in LIFE scores, with scores increasing from 2009A to 2012A. The change in status is likely due to an increase in the numbers of sensitive taxa, with this possibly also influencing the LIFE scores, although this is difficult to verify. Again, scores were generally in the same range and indicative of medium/high streamflows.

As previously mentioned, drainage, and precipitation are key factors associated with streamflow rates, potentially leading to changes in both baseflows and high flows (Blann et al., 2009; Stahl et al., 2010; Schottler et al., 2014). However, most of the studies that have so far utilised the LIFE metric, have employed it to assess drought and abstraction pressures only, within a river system (e.g. Extence et al., 1999; Bradley et al., 2017; Westwood et al., 2017). In contrast, studies aimed at assessing high flow pressures utilising the LIFE metric are relatively rare, although Dunbar et al. (2010), who demonstrated its use for assessing both high and low flows, is an exception to this.

In Ireland, drainage works are normally undertaken by the land owner. However, with the exception of schemes that are: greater than 15 ha; are likely to impact on the environment; or that occur in proposed or designated areas; registration for their implementation and/or screening is not required (Paul et al., 2018). This makes accurate recording of the prevalence of drainage difficult. Mockler et al. (2013) estimates that there is a high likelihood that 29% of land in the Republic of Ireland is under drainage, with 44% of Irelands' land in agricultural usage. Counties Clare, Galway, Leitrim, Mayo, Roscommon and Sligo, from which field work for this study was carried out, are estimated to have 34%, 19%, 51%, 22%, 44% and 27% of their land under drainage, respectively (Mockler et al., 2013). Based on visual inspections of land surrounding the sample sites, forty-seven of the sixty-five study sample sites have an associated land use type that is either agriculture or forestry related.

Despite the likelihood of drainage in the study region, and with limited trends in rainfall patterns over the twenty year period 1998-2018, only two of the twenty hydrometric stations displayed a significant increasing trend in annual mean base-flow. Similarly, only three stations in the winter and summer months displayed increasing trends for the Q95 percentile, which indicated an increase in low flows, while one station in winter had an increase in the high flow parameter Q10. Four stations did show a significant decreasing trend in annual minimum base-flow, which perhaps relates to a decrease in summer flows caused by drainage as suggested by Blann et al. (2009), and decreasing trends in the Q95 percentile were observed at three stations for the period DJFM and two stations for JJAS. The increasing streamflow trends in winter and decreasing trends in summer imply that some driver or activity is potentially impacting on some sites, while the decreasing winter streamflow trends and increasing summer trends are more difficult to explain. Unfortunately, at the regional scale it is difficult to assess accurately how streamflow changes recorded at hydrometric stations are related to changes in status at upstream or nearby sites. For example, one station with an increasing mean base-flow (station 25030) is associated with two Gained, three Lost and three Maintained sites, while the other station (29071) is associated with two Lost and two Maintained sites. Similarly, stations with a decreasing minimum base-flow trend are again associated with a mixture of status categories. Most of the sample sites from this study do not have hydrometric gauging stations located on their water body, and rarer still within range of the sampling locations, thereby allowing for only general regional hydrometric trends to be determined. Several studies have demonstrated methods for determining streamflow regimes for ungauged river sites (e.g. Ahiablame et al., 2013; Solans and Mellado-Díaz, 2015), and perhaps these methods should be employed in any future analysis to provide more detailed flow estimates. See also Mockler et al. (2013) for additional considerations from an Irish perspective.

Another limitation of this study was the relative isolation of sampling data. In contrast to analysis carried out by, for example, Westwood et al. (2017), Bradley et al. (2017), Monk et al. (2012), Dunbar et al. (2010), in which data-sets covering up to 30 consecutive years of LIFE scores were analysed, only two (consecutive) years of sampling data, with an additional two (non-consecutive) years of historical data was available for analysis here.

Seasonal and annual differences in LIFE scores were observed in this study, with Spring scores being higher than Summer scores for both years. Additionally, Figure 3.4 displays clear seasonal changes in abundances of taxa occurring based on their flow preferences. Life cycle strategies are potential key drivers of these seasonal differences with for example, *Rhithrogena* sp. which was present in high numbers during Spring samples, being almost completely absent from Summer samples, while *Leutra inermis* and *Leuctra fusca* which belong to the flow groups one and two respectively, seemingly replacing each other from Spring to Summer. In contrast, Poole et al. (2013) did not find LIFE scores to vary between Spring and Autumn in an assessment of a UK catchment, while Suren and Jowett (2006), who found seasonal differences in invertebrate densities in a New Zealand river, related the differences to impacts of streamflow, rather than life cycle strategies. Assessment of the annual differences may be best served in the context of a more extensive (temporal) dataset.

Future climate predictions for Ireland indicate river streamflows are likely to increase in winter and spring, and decrease in summer and autumn, although the magnitude of impacts may be catchment specific (Murphy and Charlton, 2006; Hall and Murphy, 2010). Predictions by Steele-Dunne et al. (2008), which included the Moy catchment that contains several rivers from this study, indicate streamflow increases of 20% in Winter and decreases of 60% in Summer between 2012 and 2060 (but see Hall and Murphy, 2010 for specific Moy Catchment predictions). Similarly, and more recently, Roudier et al. (2016) has predicted that, along with an increasing frequency of floods and extreme flows, droughts in Ireland, as well as several other European countries, are likely to become more intense as a result of climate change and a +2°C rise in temperatures. Contrastingly however, Laizé et al. (2014) places Ireland in a low/medium risk group for ecological risk due to flow alteration. Here, group six taxa, which are taxa associated with drought conditions (Extence et al., 1999) were completely absent from all sites, while taxa associated with slow flowing and standing waters, groups four and five respectively, were also relatively scarce. This indicates that none of the sites sampled here are at present susceptible to water abstraction pressures (or similar associated pressures). However, based on climate predictions this may be a future concern. Scotland, for example, which has a similar location and climate to Ireland, is predicted to have increased summer droughts as a result of climate change with potential implications for water management (Gosling, 2014). It is worth noting that a new index, the ‘Drought Effect of Habitat Loss on Invertebrates’ (DEHLI) index, especially aimed at assessing the impacts of drought on river systems has been developed (Chadd et al., 2017). This index, in contrast to the LIFE index, categorises invertebrates based on key stages of drought occurrence, and is more sensitive at picking out trends such as: the “ramp” effect – an increasing scale of river drying (and invertebrate response) with brief intermittent rainfall respites; and recovery to pre-drought state, that may otherwise be missed using the LIFE index. Perhaps in line with this but at the other end of the spectrum, a new index should also be developed that is more specific to increases in streamflow, and for example, the impacts of drainage on a riverine system.

Finally, although not discussed here, the interaction of multiple stressors, such as streamflow and sedimentation, has the potential to alter invertebrate communities, and, therefore, impact on LIFE scores (Buendia et al., 2014; Lange et al., 2014; Gieswein et al., 2017; Growns et al., 2017). A more holistic approach may, therefore, include assessments for as many stressors as possible, e.g. sedimentation (PSI scores - Extence et al., 2013; CoFSI scores - Murphy et al., 2015), organic pollution (BMWP – Hawkes, 1998), to disentangle the potential impact that each stressor has on the other.

3.5. Conclusions

In this study of Irish high status rivers with Maintained, Lost and Gained status categories, despite differences found in LIFE scores between the status categories Lost and Maintained, all scores were generally in the same range and indicative of rivers hosting invertebrate communities with a preference for medium/high streamflow rates. The historical data-sets indicated that there was no difference in LIFE scores for sites that Lost status between the 2009 and 2012 sampling periods, there-by implying streamflow pressures were not a factor in the deterioration of these sites. Again, LIFE scores were generally in the same range and associated with medium/fast streamflows. Some hydrometric stations did display changing streamflow trends, which may potentially be linked to drainage and/or change in status, although this is difficult to verify. Based on the observed invertebrate communities in this study, abstractions and/or droughts are not currently a pressure at any site. However, this may change if future climate change predictions are realised. The overall conclusion is that for most sites streamflow alterations are not likely to have been a major factor leading to deteriorations to date. However, for certain sites, and potentially in combination with other stressors, changes in streamflow patterns may be problematic.

Chapter 4.

4. Assessing the impact of fine sediment on high status river sites in Ireland

4.1. Introduction

Degradation of freshwaters resulting from excess inputs of sediment is a global concern (Richter et al., 1997; Malmqvist and Rundle, 2002; Dudgeon et al., 2006), with studies from New Zealand (Townsend et al., 2008; Ramezani et al., 2016), United States (Rabení et al., 2005), United Kingdom (Extence et al., 2017), Canada (Benoy et al., 2012) , Ireland (Conroy et al., 2016a) and Spain (Buendia et al., 2013) highlighting its impacts on aquatic biota. In the US for example, excessive sedimentation occurs in 15 % of river and stream length (USEPA, 2016). While some sedimentation, outside of the influence of human activity does occur, for example naturally occurring soil erosion of stream-banks and plays an important role in freshwater systems (Buendia et al., 2013; Turley et al., 2014), this is greatly exacerbated by anthropogenic actives (Waters, 1995; Richter et al., 1997).

Land use practices, particularly those associated with agriculture are a major contributor of excessive sediments to surface waters (Collins and Anthony, 2008; Benoy et al., 2012; dos Reis Oliveira et al., 2018). Thompson et al. (2014) suggests that for two Irish catchments, anthropogenic and agricultural activities were a key factor in the mobilising of sediments that resulted in sediment levels exceeding threshold guideline values. The main agriculture sources of sedimentation relate to: soil erosion resulting from mismanaged land, especially in relation to arable cultivation practices such as row-cropping and grazing of riparian areas by livestock (Waters, 1995; Benoy et al., 2012). Conroy et al. (2016b) and O’Sullivan et al. (2019) additionally highlight the potential for cattle accessing water-bodies as a sediment source through disturbance. While measures such as contour ploughing and fencing-off waterways should limit these threats, sedimentation still remains a major ecological concern (Matthaei et al., 2006; Sutherland et al., 2010; Bilotta et al., 2012; Sutherland et al., 2012; Glendell et al., 2014; Ramezani et al., 2014). Forestry, especially in relation to logging roads constructed close to streams, mining, and the erosion of land left un-vegetated, and urbanisation through land development, are also major sources of sedimentation (Waters, 1995; Al-Chokhachy et al., 2016; Collins and Anthony, 2008).

Fine sediment may have detrimental consequences for the ecological communities present in a water body, impacting on primary producers, invertebrates and fish (Wood and Armitage, 1997; Collins et al., 2011; Jones et al., 2012). Piggott et al. (2012) for example, found sedimentation to be the most prevalent stressor to aquatic invertebrates, in a comparison with nutrient enrichment and increased temperatures. In an experiment manipulating the addition and removal of sediment to intensive agricultural land, Ramezani et al. (2014) found both invertebrates and fish responded negatively to the addition of sediment and positively to its removal. Within a fluvial system sediment occurs as either suspended sediment floating in the water column or as deposited sediment that covers the benthic surface, although given the nature of movement within a water column, there is some degree of transfer between both types (Benoy et al., 2012). The primary impact of fine or suspended sediment on macrophytes and algae occurs through the impeding of light from penetrating the water column, which may alter the ability for periphyton and submerged and/or emergent plants to carry out photosynthesis (Bilotta and Brazier, 2008). For invertebrates, impacts occur either directly through: abrasions, clogging up of respiration mechanisms, smothering/burial, and clogging up niches in the river-bed, or indirectly through the altering of macrophyte and algal communities (Wood and Armitage, 1997; Jones et al., 2012; Extence et al., 2013). Similarly for salmonid fish, key impacts include abrasions and blocking of gill mechanisms, along with the smothering of respiring eggs/larvae (Bilotta and Brazier, 2008). The impacts of sedimentation are related to the particle size, which in turn determines whether the sediment is suspended in the water or deposited in the substrate (Waters, 1995; Wood and Armitage, 1997; Sutherland et al., 2012). To this end, many studies that assess sediment pressures tend to focus on particle sizes of either less than 0.6 mm (Glendell et al., 2014) or less than 2 mm (Zweig and Rabeni, 2001; Von Bertrab et al., 2013), as particle sizes below 2 mm are considered most harmful to aquatic biota (Waters, 1995; Ramezani et al., 2014). The amount of sediment entering a water body (Suttle et al., 2004), and the duration of sediment exposure (pulses) (Shaw and Richardson, 2001) are also important considerations. Along with the afore-mentioned factors, Bilotta and Brazier (2008) additionally highlight how the effect on aquatic biota may vary depending on the chemical composition of suspended sediment, and its potential to alter the chemical composition of receiving waters.

Aligned to this is the interaction of “multiple stressors” (Matthaei et al., 2010; Piggott et al., 2012; Lange et al., 2014a). Turley et al. (2016) provides a summary of confounding pressures associated with sedimentation and their impact on invertebrates that includes flow, nutrients, pesticides, metals and pathogens. A reduced flow, from for example, increased water abstraction, may increase the amount of sedimentation in, and temperature of, a stream, as well as altering dissolved oxygen (DO), pH and nutrient levels (Dewson et al., 2007). Sedimentation may also be increased by high flows, as Beckmann et al. (2005) reports that current velocity in the tributary mouths of the River Rhine, decreases by 40-50% during high flow, which potentially increases the levels of fine sediment in the water column.

High status water-bodies (HSWs) are rivers, lakes, transitional and coastal waters, that are defined under the European Union Water Framework Directive (OJEC, 2000) as being close to reference conditions, based on a limited/minimal influence from anthropogenic activities (WG 2.3, 2003; Mayes and Codling, 2009). Relative to other EU countries, Ireland has a high number of HSWs (data extracted from the Europe (WISE) - WFD database - EEA, 2015). However, in recent years large deteriorations have been observed (EPA, 2012; EPA, 2016; White et al., 2014), and along with nutrient enrichment, flow modifications and pesticide/herbicide usage, these deteriorations have potentially been attributed to increasing levels of sedimentation (Ní Chatháin et al., 2012; White et al., 2014). For example, increased sedimentation is cited as a key factor associated with declines in the Freshwater Pearl Mussel (*Margaritifera margaritifera*) (Leitner et al., 2015; Gumpinger et al., 2015), a species very often associated with HSWs. It is this relationship between HSW deteriorations and increasing sedimentation levels that was investigated in this study.

Invertebrates have routinely been used for assessing water quality degradation because of, for example, 1) the relative ease of sampling, 2) a sensitivity to various pollution stressors and habitat modifications especially related to streamflow rates and siltation, 3) a variation amongst taxa of tolerance/sensitivity levels which allows for a scoring system, such as the Biological Monitoring Working Party (BMWP) (Hawkes, 1998), to be utilised, and 4) dichotomous keys are available for most groups (Hellawell, 1986; Rosenberg and Resh, 1993; Zalack et al., 2010). Additionally, they represent a middle trophic ground, between primary producers (algae) and top end predators (fish) (Relyea et al., 2011), and their use is a key requirement of the WFD (OJEC, 2000). In line with this, recent efforts for assessing the impacts of sedimentation have focused on the use of invertebrates. For example, Relyea et al. (2011) developed, using historic data-sets, the Fine Sediment Biotic Index (FSBI) to assess the impact of fine sediment (< 2mm) on North-western United States streams. In the UK metrics have been developed using a literature review (Extence et al., 2013), empirical evidence (Murphy et al., 2015) or a combination of both Turley et al. (2015).

With this background, and with regard to the deterioration of HSWs in Ireland, the aim of this study was to examine sedimentation as a pressure on high status river biology. The objectives were to: 1) use invertebrates and sediment specific indices as developed by Extence et al. (2013), Murphy et al. (2015) and Turley et al. (2015; 2016), to assess the impact of sedimentation on river biology; 2) assess the relationship between physical sediment variables and the change in status of HSW rivers; 3) assess using a historical data-set, the relationship between sediment pressures and change in status of HSW rivers; and 4) assess the relationship between sedimentation and other stressors (streamflow alteration and nutrient enrichment) in HSWs.

4.2. Materials and Methods

4.2.1. Macro-invertebrate Sampling

Macro-invertebrates were collected on four sampling occasions from sixty-five high status river sites, using a three-minute kick-sample, followed by a one-minute stone searching, as described by methods in BS-ISO (2012) and Environment Agency (2012). The macro-invertebrate samples were preserved in 75 % alcohol on the day of collection. The sample dates were in April/May (Spring) and August (Summer) in 2016 and 2017. The sixty-five sample sites were selected at random from 167 high status river sites in the west of Ireland that were initially coded for slope and hardness based on the RIVtypes classification (Kelly-Quinn et.al., 2005) (Figure 4.1) (see also Appendix B for sample site details). Using data obtained online from the Environmental Protection Agency (EPA) ecological quality (Q-value) reports (http://epa.ie/QValue/webusers/), which are primarily generated using macro-invertebrates, the sixty-five sites were categorised as having either: Lost their high status (e.g. gone from high to good, moderate, poor or bad); consistently Maintained high status; or had Gained in status (e.g. from good to high) (see Appendix B). This resulted in the selection of 20 Lost sites, 24 Maintained sites and 21 Gained sites. Poor sampling conditions excluded three sites from each of the Summer 2016 and Summer 2017 sampling programmes.

4.2.2. Sediment metrics – PSI, CoFSI and E-PSI scores

Macro-invertebrates were identified to the lowest practical taxonomic level using an Olympus (SZX16) stereo microscope and relevant dichotomous keys (a list of keys is provided in Appendix G). This was generally to species or genus level, with the

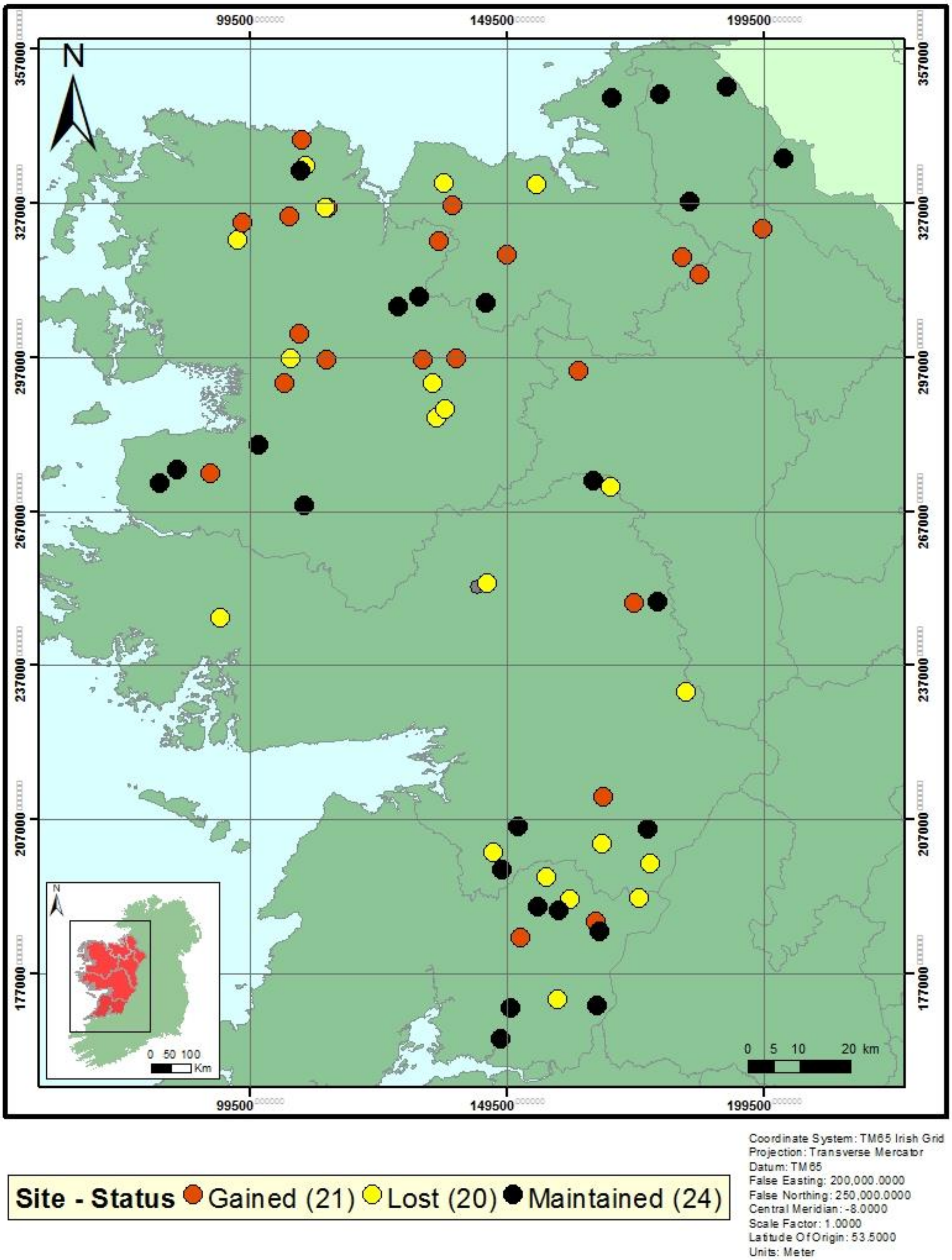


Figure 4.1. Location of the sixty-five sampling sites that are categorised as having either: lost their high status (e.g. gone from high to good, moderate, poor or bad); consistently maintained high status; or gained in status (e.g. from good to high).

exception of Oligochaetes (Order) and Dipterans (Family or Tribe). Following identification, sediment specific biotic metrics were applied.

The Proportion of Sediment-sensitive Index (PSI) as described by Extence et al. (2013) assesses the impact of fine sedimentation deposition on lotic ecosystems using a macro-invertebrate scoring system, developed by carrying out an extensive literature review, as well as assessing the physical and physiological characteristics of invertebrate taxa, relative to sediments (Extence et al., 2013). Invertebrate taxa (either to species or family level) are assigned to groups A, B, C and D depending on their sensitivity to sediment levels, with these groups representing: highly sensitive, moderately sensitive, moderately insensitive and highly insensitive, respectively. The PSI score also takes account of abundances and is calculated as:

$$PSI(\Psi) = \frac{\sum Scores\ for\ Sediment\ Sensitivity\ Groups\ A\&B}{\sum Scores\ for\ Sediment\ Sensitivity\ Groups\ A,B,C\ \&\ D} \times 100 \quad \text{[Eq. 4.1]}$$

where taxa are assigned to a group determined from a list of taxa in the Appendix of Extence et al. (2013), and scores are generated based on a combination of the taxa's assigned group and abundance category at which it occurs (i.e. Table 4.1 from Extence et al. (2013)). Extence et al. (2013) also provides a table for interpreting the generated PSI scores (Table 4.1).

Table 4.1 Interpretation of PSI scores (Extence et al., 2013).

PSI	River bed condition
81–100	Minimally sedimented/unsedimented
61–80	Slightly sedimented
41–60	Moderately sedimented
21–40	Sedimented
0–20	Heavily sedimented

Monk et al. (2012) highlight the benefits of using species over families when applying the Lotic-invertebrate Index for Flow Evaluation (LIFE) index, with for example, taxa within families having different flow type preferences. The same reasoning applies to the PSI, where again taxa within some families are associated with different sediment groups. Furthermore, greater taxonomic resolution should enable for the identification of the impacts of invasive species on scoring metrics (Mathers et al., 2016).

Additionally, the Combined Fine Sediment Index (CoFSI) which was developed by Murphy et al. (2015), with the aid of empirical evidence and multivariate ordination techniques, was employed to assess the impacts of sedimentation on invertebrates. The CoFSI index assigns an organic Fine Sediment Index (oFSI) score out of ten and a Total Fine Sediment Index (ToFSI) score out of ten to a list of one hundred and five taxa, with a score of zero being sediment tolerant and ten being sediment sensitive. The oFSI and ToFSI scores are then combined to give a total CoFSI score using:

$$CoFSIsp = 0.349\ (oFSIsp) + 0.569\ (ToFSIsp) \qquad \text{[Eq. 4.2]}$$

The higher the CoFSI score the greater the sensitivity to sedimentation of the invertebrate community.

A third metric for assessing sedimentation, the Empirically-weighted PSI (E-PSI) as developed by Turley et al. (2016) was also employed in this study. The E-PSI metric combines elements of the PSI metric with optimal weightings extracted from an empirically generated training data-set (see Turley et al., 2016, for weighting scores). Invertebrates are classified as either sensitive or insensitive to sedimentation, and within these categories empirically derived weightings are applied. The metric is calculated as:

$$E - PSI = \frac{\sum(logAsens \times W)}{\sum(logAll \times W)} \times 100 \qquad \text{[Eq. 4.3]}$$

where the sum of the log of the abundance of sensitive taxa ($logA_{sens}$) multiplied by its associated weighting, is divided by the sum of the log of the abundance of all the taxa (sensitive and insensitive combined) ($logA_{all}$) multiplied by the associated weighting. The result is then multiplied by 100 to give the E-PSI score. For this metric, the log abundance categories were generated as: 1-9 individuals = 1; 10-99 = 2, 100-999 = 3 and 999+ = 4 (Turley et al., 2016). In this study mixed taxon/species level E-PSI scores were generated. Again, higher E-PSI scores are associated with reduced sedimentation pressures.

Other metrics calculated in this study include the Biological Monitoring Working Party (BMWP) (Hawkes, 1998) and Whalley, Hawkes, Paisley and Trigg (WHPT) (WFD-UKTAG, 2014) and their associated Average Score Per Taxon (ASPT) and Number of Scoring Taxa (NTAXA), all of which were calculated at family level. These metrics are generally employed to provide an assessment of general/organic pollution within a waterbody. A summary of the indices used in this study are presented in Table 4.2.

Using the R software programme (R Core Team, 2018), Wilcoxon-Mann-Whitney tests, were used to test for differences between the PSI scores of different status categories, e.g. Lost against Maintained, within each sample period. Wilcoxon-Mann-

Whitney tests, were used as the data were not normally distributed and were non-transformable, and the datasets were independent of each other. Wilcoxon-Mann-Whitney tests were also used to test for direction of differences between categories (i.e. greater or less than). Wilcox Signed Rank tests were used to test for differences in PSI scores between seasons and also between years, as datasets in this case were paired. This was also repeated within each status category, and again the direction of change was analysed. Similar data analysis was repeated for CoFSI, E-PSI, BMWP, ASPT (BMWP), NTAXA (BMWP), WHPT, ASPT (WHPT), and NTAXA (WHPT) scores. Some caution is required with the interpretation of these results as both the classification of status categories, and the generation of for example, PSI scores, employ invertebrates, and so are not therefore, fully independent of each other. However, while the generation of status categories through the EPA Q-value system is more aimed at assessing general/organic pollution patterns, the generation of, for example, PSI, CoFSI and E-PSI scores are specifically related to the sensitivity of invertebrate taxa to sedimentation pressures.

Table 4.2. Summary table of indices used in this study, their purpose and the associated reference.

Index	Purpose	Reference
Proportion of Sediment-sensitive Index (PSI)	Sediment index	Extence et al. (2013)
Combined Fine Sediment Index (CoFSI)	Sediment index	Murphy et al. (2015),
Empirically-weighted PSI (E-PSI)	Sediment index	Turley et al. (2016)
Biological Monitoring Working Party (BMWP)	General/organic pollution	Hawkes (1998)
Whalley, Hawkes, Paisley & Trigg (WHPT)	General/organic pollution	WFD-UKTAG (2014)
Average Score Per Taxon (ASPT)	Organic pollution	Assoc. with BMWP and WHPT
Number of Scoring Taxa (NTAXA)	Non-specific, toxins, habitat	Assoc. with BMWP and WHPT
Lotic Invertebrate Flow Evaluation (LIFE)	Flow sensitivity	Extence et al. (1999)

4.2.3. Generation of historical PSI scores

Using EPA monitoring data for the sampling periods 2007, 2008 and 2009 (labelled 2009A) and for the sampling periods 2010, 2011 and 2012 (labelled 2012A), historical PSI scores were generated for 286 of the high status river sites recorded through-out Ireland (Figure 4.2). EPA monitoring of macro-invertebrates using the Q-value system identifies macro-invertebrates as occurring as either Present (1-2 individuals), Scarce/Few (<1% of the total sample), Small numbers (<5%), Fair numbers (5-10%), Common (10-20%), Numerous (25-50%), Dominant (50-75%) and Excessive (>75%) (McGarrigle et al., 2002). However, within the dataset for the 286 sites, only the abundance categories "single, few, common, numerous and dominant" were present. Unmodified, these EPA abundance categories are incompatible with the PSI metric, which categorises invertebrate abundances within a log scale, i.e. of 1-9, 10-99, 100-999, 1000-9999, and <10,000 (Extence et al., 2013). Therefore, in order to generate PSI scores from the EPA datasets (and incorporating as a harmonising assumption), the EPA abundance categories were re-assigned as: Single and Few = 1-9; Common = 10-99; Numerous = 100-999; and Dominant = >999.

As a check on this assumption, actual PSI score data, generated from the Summer 2016 sampling period, were compared with a PSI score that was generated using the same invertebrates, that were given as a percentage of the total invertebrate count for that sample. The percentage abundance was then assigned as either Single, Few, Common etc., depending on the percentage category. The percentage categories were: 0 > Few < 9% ; >=10 % Common =< 22% ; >= 23% Numerous <= 50% ; > 50% Dominant <= 75% ; and > 75% = Excessive. These categories were then assigned as, for example, Few = 1-9, Common = 10-99, etc. A Spearman rank correlation of the actual PSI scores against the generated PSI scores had a correlation coefficient of 0.972 at $p<0.01$, indicating a very strong relationship. A similarly strong relationship (0.999 at $p<0.01$) was observed when the generated PSI score was created by directly assigning the categories Few, Common, etc. based on the numbers of taxa present, without first calculating the abundance percentages. Plots of the correlations of actual PSI scores from Summer 2016 against the generated PSI scores from Summer 2016 are presented in Appendix D, Figure D1.

To assess the relationship between sediment pressures and change in status of HSW rivers using the more extensive EPA historical dataset, historical PSI scores for the periods 2009A and 2012A, for the 286 high status sites were compared with each other, with paired Wilcox Signed Rank tests used to analyse the data.

4.2.4. Physical assessment of fine sediment

To assess fine sediment (< 2 mm) pressures at the sixty-five river sites, five sediment assessment methods (two re-suspendable sediment and three deposited methods) were carried out. The primary re-suspendable sediment analysis method employed was the "Quorer" method, adapted from methods described by Quinn et al. (1997), Collins and Walling (2007), Clapcott et al. (2011), Glendell et al. (2014), Lange et al. (2014a; 2014b) and Duerdoth et al. (2015), whereby a metal bin of diameter 40 cm and height 60 cm was pushed into the river-bed sediment to a depth of ca. 2-5 cm, forming a seal with the river-bed substrate. Using a metre rule, the height of water within the bin was measured three times and the average height recorded. The water and upper 5 cm of the substrate within the bin was then disturbed with a metal rod for approximately 60 seconds. A 500 ml sample bottle was then immediately plunged into the bin/water to take a representative aliquot of the mobilised sediment. This process was repeated three times across the width of each river. Following collection and return to the laboratory, the 500 ml samples were stored in a fridge until it was time for analysis, at which time they were returned to room temperature. The vacuum filtration method, using 0.45 μm Whatmann glass-fibre filters was used to determine the sediment concentration Cs(t) (g/L) within the 500 ml sample bottles. Following this, the amount of fine sediment per unit surface area $B_r(t)$ (g/m^2) was determined, as described by Collins and Walling (2007), using the equation:

$$B_r(t) = \frac{C_s(t)W_v(t)}{A} \qquad \text{[Eq. 4.4]}$$

Where $W_v(t)$ (L) is the volume of water within the sampling bottle (500 ml) and A is the surface area ($2\pi rh+2\pi r^2$) of the sampling bin whose height h (m) is equivalent to the depth of water within the bin, and r is the bin radius.

A second re-suspendable sediment method "Tile", as described by Clapcott et al. (2011), involved disturbing the river bed substrate upstream of a white tile (15 cm X 15 cm) placed on the substratum, and assigning a score of one to five based on the visibility and duration of the resulting plume. A score of one was associated with no plume and a still visible white tile, while a score of five was given if the white tile completely disappeared under the resulting plume. In comparison to the Quorer method, the white tile provides a rapid qualitative assessment of the "total suspendable solids" present on the river substratum.

The deposited sediment assessment methods included two visual assessment methods and an assessment of sediment depth. The first visual assessment method "% Fine", follows that of the Environment Agency (2012) and categorises the river bed substratum as either silt/clay, sand, pebbles/gravel, or boulders/cobbles based on particle size and texture as outlined in Table 4.3. In this study, the silt/clay and sand were pooled together to give the proportion of substrate that has a particle size less than 2 mm. The second visual assessment method, the viewing box method "Scope", was modified from that described by Zweig and Rabeni (2001), Matthaei et al. (2006) and Clapcott et al. (2011). It estimates the percentage of fine sediment (< 2mm) cover of the river substrate within a 20 x 20 cm grid box drawn onto the bottom of a bathoscope. Finally, sediment "Depth" was measured as per Lange et al. (2014a; 2014b), whereby a 60 cm X 60 cm sampling frame, (constructed of copper pipe of diameter 1.3 cm, soldered at the joints), was tossed randomly (n = 3), and at the centre of the sampling frame, a metal ruler was pushed into the river bed until underlying coarser material was reached. Wilcoxon-Mann-Whitney tests, were used to test for differences between the sediment variables of different status categories, e.g. Lost against Maintained, within each sample period. Wilcox Signed Rank tests (paired datasets) were used to test for differences in sediment variables between seasons and also between years.

Table 4.3. Substratum particle size categories as recorded by the Environment Agency (2009).

Category	Width (mm)	Description
Silt/clay	<0.06	Soft in texture and not abrasive to the hands when rubbed.
Sand	0.06 - 2	Smaller than instant coffee granules and, unlike silt/clay, abrasive to the hands when rubbed.
Pebbles/gravel	2 - 64	Instant coffee granule to half fist size.
Boulders/cobbles	>64	Half fist size or larger.

4.2.5. Physico-chemical assessment

pH (Spring 2017 only) and conductivity, and temperature and dissolved oxygen (excluding Spring 2016) readings were taken *in situ* on the same dates as invertebrate sampling using portable Hach meters. Additionally, at each site, 50 ml filtered water samples were collected to determine soluble reactive phosphorus (SRP). The water samples were filtered using a 50 mL syringe and a Polyethersulfone (PES) membrane filter, of pore size 0.45 μm, into two 50 ml high density plastic bottles. The SRP was determined on the day of sampling using a Hach DR2800 portable spectrophotometer, and the phosphomolybdate method. Additionally, during the Summer 2016 sampling period, a 2 L sample of river water was collected at each site. These 2 L samples were delivered to the EPA laboratories in Castlebar at the end of each day and analysed for a suite of physical-chemical components, including: biochemical oxygen demand (BOD) (range of measurement 1-100,000mg/l), ammonia (0.02 – 10,000 mg/l N), total oxidized nitrogen (TON) (0.02 – 5,000 mg/l N) , nitrate (0.02 – 5,000 mg/l N), nitrite (0.004 – 50 mg/l N), o-phosphate (0.01 – 1,000 mg/l N), chloride (2 – 50,000 mg/l), alkalinity (10 – 10,000 mg/l $CaCO_3$) and hardness (10 – 10,000 mg/l $CaCO_3$). All methods follow the EPA W07 standard operating procedures except BOD (EPA W04), alkalinity (EPA W17) and hardness (EPA W16), as described in Irish National Accreditation Board (2019). Samples were analyzed within 24hrs of collection.

4.2.6. Spearman rank correlations.

Non-parametric Spearman rank correlation tests were conducted between each of the biological metrics and each of the sediment variables for each sampling period using SPSS version 23 (IBM, 2015) as the datasets were non-normally distributed and un-transformable. An additional biological metric, the Lotic Index for Flow Evaluation –

(LIFE) metric as developed by Extence et al. (1999), that is employed to assess the relationship between flow and invertebrate communities, was also included in the Spearman Rank analysis so as to assess the relationship between flow and sedimentation. See Chapter 3 (Flow Chapter) for the generation and analysis of the LIFE scores used in this study. As one of the assumptions of the Spearman Rank correlation test is for a monotonic relationship between the two variables being tested, and this was not evident between every tested set of variables, some degree of caution should be used when interpreting these results.

4.3. Results

4.3.1. Invertebrate communities present

For each sample site, and each sample season the majority of individual invertebrate taxa types present, were taxa that are either highly sensitive (Group A) or moderately sensitive (Group B) to sedimentation (Figures 4.2 and 4.3). This was similar for each of the Lost, Gained and Maintained status categories. For Spring of both 2016 and 2017 highly sensitive taxa were the most abundant across all status categories. However, during Summer 2016 and 2017, moderately sensitive taxa were the most abundant (Figure 4.3).

Baetis rhodani (65 sites), *Alainites (Baetis) muticus* (57 sites), *Rhithrogena semicolorata* (56 sites), *Hydropsyche siltalai* (53 sites), *Isoperla grammatica* (46 sites), *Siphonoperla (Chloroperla) torrentium* (45 sites) *Leuctra inermis* (42 sites), *Ecdyonorus* sp. (39 sites) and *Agapetus* sp. (41 sites) were the most commonly occurring Group A taxa in Spring 2016. Of these, *Baetis rhodani* and *Rhithrogena semicolorata* were the most abundant. *Limnius volckmari* (63 sites), *Elmis aenea* (62 sites) and *Gamarus dubini* (62 sites) were the most commonly occurring and abundant Group B taxa. Of the moderately insensitive taxa (Group C), *Esolus parallelepipedus* (58 sites), *Caenis rivulorum* (43 sites), *Potamopyrgus antipodarum* (31 sites) and *Oulimnius tuberculatus* (30 sites) were the most commonly occurring and abundant, while Oligochaeta (61 sites) were the most commonly occurring and abundant very insensitive (Group D) taxa. For all Groups A-D, the most commonly occurring and most abundant taxa, as outlined above for Spring 2016 was similar in Spring 2017.

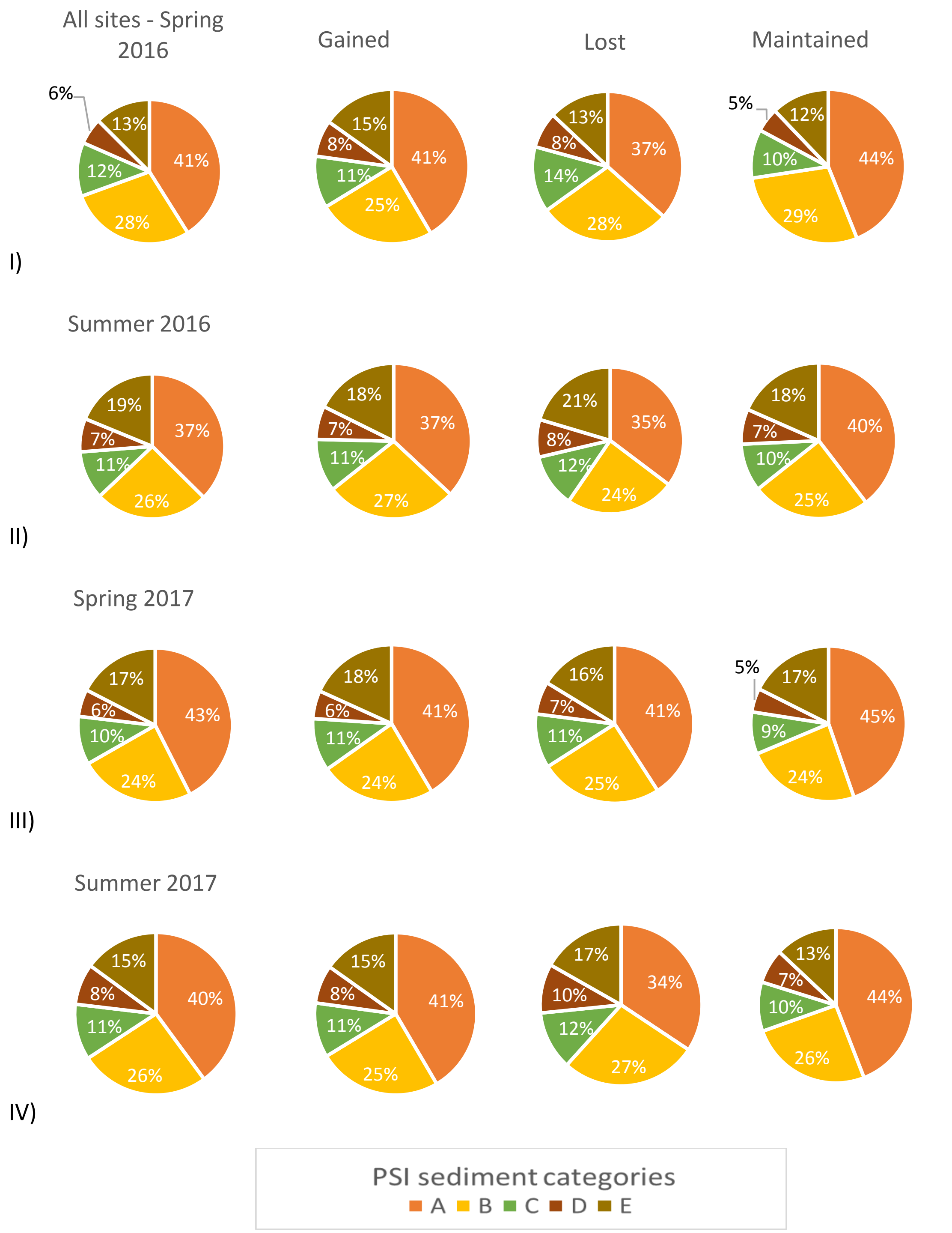


Figure 4.2. Average number of individual taxa types occurring in the sixty-five samples sites based on their PSI defined sediment sensitivity, where: A) is highly sensitive; B) is moderately sensitive; C) is moderately insensitive; D) is highly insensitive; and E) is excluded from PSI scoring; from I) Spring 2016; II) Summer 2016; III) Spring 2017; and IV) Summer 2017.

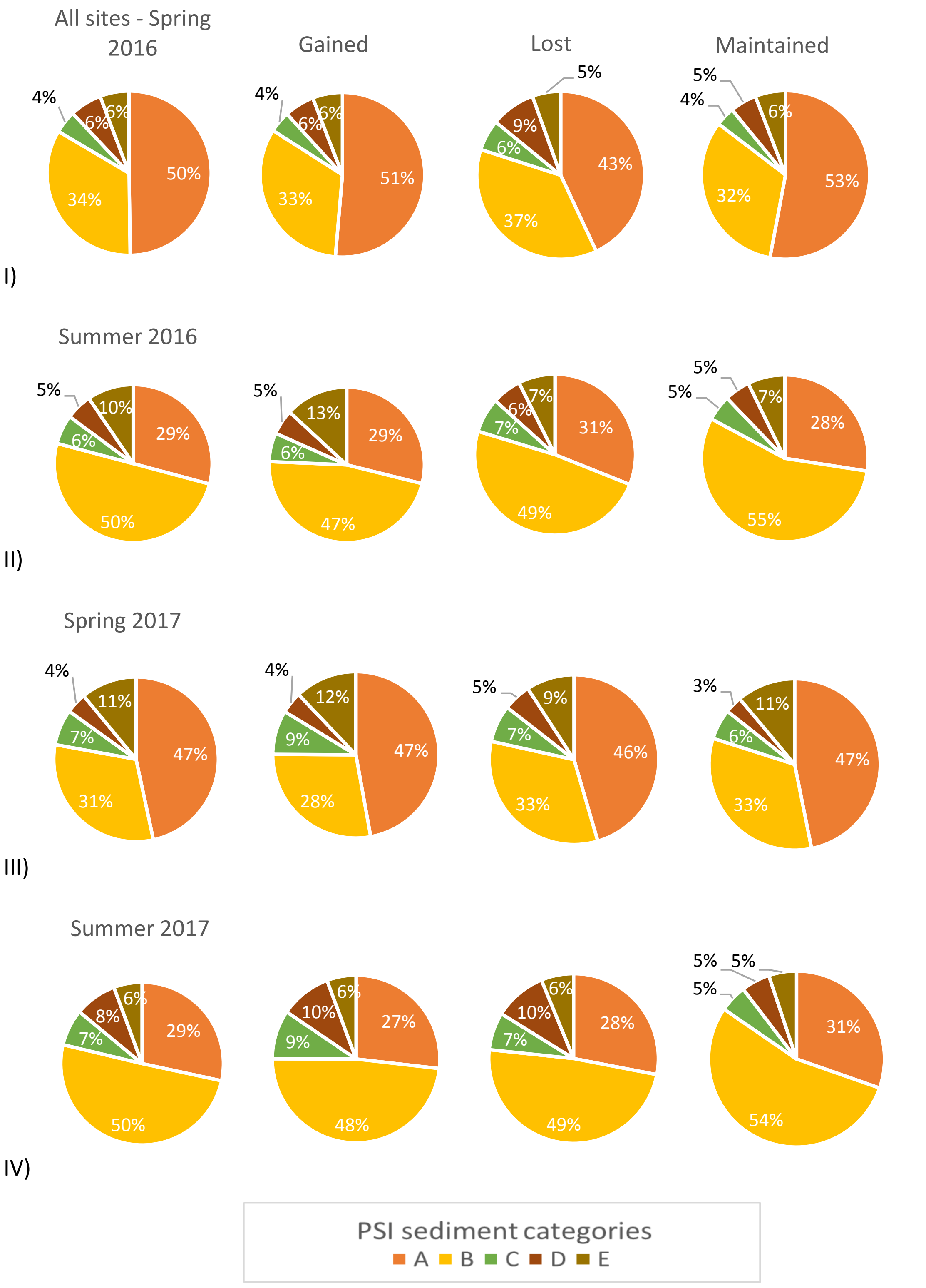


Figure 4.3. Abundances of taxa occurring in the sixty-five samples sites based on their PSI defined sediment sensitivity, where: A) is highly sensitive; B) is moderately sensitive; C) is moderately insensitive; D) is highly insensitive; and E) is excluded from PSI scoring; from I) Spring 2016; II) Summer 2016; III) Spring 2017; and IV) Summer 2017.

In Summer 2016, *Baetis rhodani* (61 sites), *Leuctra fusca* (58 sites), *Alainites muticus* (57 sites), *Serratella ignita* (53 sites), *Ecdyonurus* sp. (50 sites), *Rhyacophila dorsalis* (46 sites) and *Hydropsyche pellucidula* (34 sites) were the most commonly occurring and among the most abundant Group A taxa. As with Spring 2016, *Limnius volckmari* (60 sites), *Elmis aenea* (61 sites) and *Gamarus dubini* (62 sites) were again the most commonly occurring and abundant Group B taxa in Summer 2016. The most commonly occurring and most abundant taxa, as outlined above for Groups A and B in Summer 2016, was similar in Summer 2017. The Group C and D taxa listed above for Spring 2016, were again the most commonly occurring and abundant taxa in Summer 2016 and 2017.

4.3.2. PSI, CoFSI and E-PSI Scores (2016 and 2017)

The average PSI scores for each status category for each sample period, and the average number of scoring taxa for each status category are presented in Tables 4.4 and 4.5 (see also Appendix D for full list of PSI, CoFSI, E-PSI, BMWP, WHPT and ASPT scores). Across all sampling periods, the majority of sites were either minimally sedimented/unsedimented (i.e. PSI scores between 81-100) or slightly sedimented (PSI scores 61-80) (Tables 4.6 and 4.7). Only one site, (34C100300 – Lost status) had a PSI score that classified it as sedimentated, although this site fluctuated between slightly sedimented and moderately sedimented by Summer 2017. Between Spring and Summer for both years, the number of sites that were minimally sedimented/unsedimented decreased, while in contrast, the number of slightly sedimented sites increased (Table 4.6 and 4.7).

Table 4.4. The average PSI, E-PSI and CoFSI scores for each status category for each sample period, with standard deviation in parenthesis.

	Spring 2016			Summer 2016		
	PSI	**CoFSI**	**E-PSI**	**PSI**	**CoFSI**	**E-PSI**
All	81.62 (9.4)	126.27 (22.5)	94.56 (7.0)	80.47 (7.5)	111.05 (23.0)	93.88 (5.7)
Gained	82.79 (4.9)	131.75 (21.5)	95.52 (3.7)	80.44 (5.6)	113.42 (25.9)	93.88 (4.0)
Lost	77.05 (12.3)	115.52 (23.6)	91.42 (10.3)	77.82 (6.6)	105.84 (24.5)	92.2 (6.4)
Maintained	84.37 (8.3)	130.2 (19.2)	96.34 (4.7)	82.57 (8.8)	112.96 (17.8)	95.18 (6.0)

Table 4.5. The average PSI, E-PSI and CoFSI scores for each status category for each sample period, with standard deviation in parenthesis

	Spring 2017			Summer 2017		
	PSI	**CoFSI**	**E-PSI**	**PSI**	**CoFSI**	**E-PSI**
All	83.12 (6.3)	137.87 (27.1)	95.8 (3.9)	79.39 (8.6)	101.61 (24.4)	93.06 (6.1)
Gained	82.48 (5.8)	143.72 (22.3)	95.42 (3.8)	80.03 (7.1)	104.68 (29.9)	93.58 (5.4)
Lost	81.3 (7.0)	125.99 (34.1)	94.54 (4.1)	75.57 (9.9)	101 (21.9)	90.53 (6.9)
Maintained	85.19 (5.6)	142.66 (20.6)	97.19 (3.4)	82.27 (7.1)	99.37 (20.5)	94.9 (5.1)

Table 4.6. Number of sites with PSI scores as per Extence et al. (2013) of: 81-100 - Minimally sedimented/unsedimented; 61-80 - Slightly sedimented; 41-60 - Moderately sedimented; 21-40 - Sedimented; and 0-20 - Heavily sedimented; for each status category (Gained, Lost and Maintained) for Spring and Summer 2016.

Interpretation of PSI scores	**Spring 2016**				**Summer 2016**			
PSI River bed condition	**Gained**	**Lost**	**Maint.**	**Total**	**Gained**	**Lost**	**Maint.**	**Total**
81–100 Min. sedim./unsedim.	16	10	19	45	11	6	16	33
61–80 Slightly sedimented	5	9	4	18	10	12	6	28
41–60 Moderately sedimented			1	1			1	1
21–40 Sedimented		1		1				
0–20 Heavily sedimented								

Table 4.7. Number of sites with PSI scores as per Extence et al. (2013) of: 81-100 - Minimally sedimented/unsedimented; 61-80 - Slightly sedimented; 41-60 Moderately sedimented; 21-40 Sedimented; and 0-20 Heavily sedimented; for each status category (Gained, Lost and Maintained) for Spring and Summer 2017.

Interpretation of PSI scores	**Spring 2017**				**Summer 2017**			
PSI River bed condition	**Gained**	**Lost**	**Maint.**	**Total**	**Gained**	**Lost**	**Maint.**	**Total**
81–100 Min. sedim./unsedim.	16	12	18	46	8	6	15	29
61–80 Slightly sedimented	5	8	6	19	12	13	7	32
41–60 Moderately sedimented						1		1
21–40 Sedimented								
0–20 Heavily sedimented								

Analysis of PSI scores found a significant difference between Maintained and Lost sites in Spring 2016, Summer 2016, Spring 2017 and Summer 2017, with p values of 0.014, 0.017, 0.043 and 0.016, respectively. For all significant differences Maintained sites had significantly greater PSI scores than Lost sites. Across all sampling periods, no significant difference in PSI scores was found between Lost and Gained sites, and Maintained and Gained sites.

Analysis over the two years of sampling found no difference between PSI scores from Spring 2016 and Spring 2017, or between Summer 2016 and Summer 2017. Seasonal analysis found a significant difference between Spring 2016 and Summer 2016 ($p = 0.014$) and between Spring 2017 and Summer 2017 ($p<0.01$), with Spring scores being greater than Summer scores for both years ($p< 0.01$). Within the Maintained category there were no significant differences between Maintained PSI scores in Spring 2016 and Spring 2017, and Summer 2016 and Summer 2017, but there were seasonal differences between Spring 2016 and Summer 2016 ($p = 0.045$), and Spring 2017 against Summer 2017 ($p= 0.013$). Spring scores were greater than Summer scores in 2016 ($p = 0.022$) and 2017 ($p< 0.01$). Within the Lost and Gained categories only seasonal differences were found: between Spring and Summer 2017 ($p=0.026$ - Lost); and Spring and Summer 2016 ($p=0.05$ - Gained). Spring scores were greater than Summer scores on each occasion.

For the variable CoFSI, statistical differences were only found in Spring 2016, between Lost and Maintained ($p = 0.041$) and Lost and Gained ($p = 0.048$), with Lost being lower on both occasions, ($p = 0.02$ and $p = 0.024$, respectively). Significant yearly differences in CoFSI scores between Spring 2016 and Spring 2017, and between

Summer 2016 and Summer 2017 were found, both at $p < 0.01$. Seasonal differences, between Spring 2016 and Summer 2016, and between Spring 2017 and Summer 2017, were also found, with both again at $p < 0.01$.

Within the Gained and Maintained status categories, there were significant differences between CoFSI values found in: Spring 2016 and Spring 2017 (Gained - $p = 0.038$; Maintained - $p = 0.029$); Spring 2016 and Summer 2016 (both $p<0.01$); and Spring 2017 and Summer 2017 (both $p<0.01$). Within the Lost status category, there was a significant difference only between CoFSI values found in Spring 2017 and those found in Summer 2017 ($p < 0.01$).

All sample sites, with the exception of site 34C100300 (Lost) in Spring 2016 (which had an E-PSI score of 55.08), had an E-PSI score greater than 70. E-PSI statistical differences were only found between Lost and Maintained, with these differences occurring in Spring 2016 ($p = 0.021$), Summer 2016 ($p = 0.034$), Spring 2017 ($p < 0.01$) and Summer 2017 ($p = 0.016$). On each occasion Maintained scores were greater than Lost scores. No yearly differences in E-PSI scores were found, although seasonal differences, between Spring 2016 and Summer 2016 ($p = 0.031$), and between Spring 2017 and Summer 2017 ($p < 0.01$), were found, with Spring scores being greater than Summer scores on each occasion. Within the Gained status category, no differences between years or between seasons were found. Within the Lost and Maintained status categories, there were significant differences between E-PSI values found in Spring 2017 and Summer 2017 (both $p<0.01$); while Spring 2017 and Summer 2017 had a p-value of 0.054 in the Maintained category; and Spring 2016 and Spring 2017 had a p-value of 0.082 in the Lost category.

Of the other biological variables, significant differences between status categories were found for the variable ASPT (BMWP) in Spring 2016, between Lost and Gained ($p = 0.013$) and between Lost and Maintained ($p = 0.023$), and in Summer 2016 between Lost and Maintained ($p = 0.013$) and Lost and Gained ($p < 0.01$). Lost scores were statistically less than those of Gained and Maintained in both Spring and Summer 2016. Additionally, for the variable ASPT (WHPT), statistical differences were found between Lost and Maintained in Spring 2016 ($p = 0.014$), Summer 2016 ($p = 0.027$) and Summer 2017 ($p < 0.01$); and between Gained and Maintained ($p = 0.048$) in Summer 2017. Maintained scores were statistically greater than Gained and Lost scores.

Significant yearly differences in BMWP scores between Spring 2016 and Spring 2017, and Summer 2016 and Summer 2017, were found, both at $p <0.01$. Seasonal differences, between Spring 2016 and Summer 2016, and Spring 2017 and Summer 2017, were also found, with both again having $p<0.01$. This was also the case for yearly and seasonal analysis of N Taxa (BMWP) scores, although with p values of 0.011 and 0.012 for Spring 2016 against Summer 2016, and Spring 2017 against Summer 2017, respectively. For ASPT (BMWP) there was no difference between Spring 2016 and Spring 2017 scores, but there was a difference between Summer 2016 and Summer 2017 ($p=0.031$). Seasonal differences occurred for both years ($p<0.01$ for both). For WHPT, N Taxa (WHPT) and ASPT (WHPT) no yearly differences between Spring 2016 and Spring 2017, or between Summer 2016 and Summer 2017, were found, although seasonal differences for each variable, for both years (all at $p<0.01$) were observed.

Table 4.8. The average WHPT scores, with corresponding NTaxa and APST scores for each status category for the sample periods Spring and Summer 2016, with standard deviation in parenthesis.

	Spring 2016			Summer 2016		
	WHPT	**Ntaxa**	**ASPT (WHPT)**	**WHPT**	**Ntaxa**	**ASPT (WHPT)**
All	160.6 (33.9)	23.2 (4.4)	6.9 (0.5)	139.3 (31.7)	21.2 (4.4)	6.6 (0.5)
Gained	162.3 (38.4)	23.2 (4.9)	7 (0.4)	146.6 (37.3)	21.9 (4.8)	6.6 (0.5)
Lost	152.2 (31.5)	22.9 (4.4)	6.7 (0.6)	129.8 (29.8)	20.4 (4.4)	6.3 (0.4)
Maintained	166.1 (29.8)	23.5 (3.7)	7 (0.5)	140.2 (25.0)	21.2 (3.7)	6.6 (0.5)

Table 4.9. The average WHPT scores, with corresponding NTaxa and APST scores for each status category for the sample periods Spring and Summer 2017, with standard deviation in parenthesis.

	Spring 2017			Summer 2017		
	WHPT	**Ntaxa**	**ASPT (WHPT)**	**WHPT**	**Ntaxa**	**ASPT (WHPT)**
All	165.7 (33.8)	23.7 (4.3)	7 (0.4)	131.7 (33.6)	20.3 (4.9)	6.5 (0.4)
Gained	171.4 (33.3)	24.6 (4.1)	6.9 (0.4)	135.1 (42.9)	20.9 (6.3)	6.4 (0.4)
Lost	154.7 (39.3)	22.3 (5.3)	6.9 (0.5)	126.3 (30.2)	20.1 (4.8)	6.3 (0.3)
Maintained	169.8 (26.3)	24 (3.0)	7.1 (0.4)	133.6 (25.1)	20 (3.2)	6.7 (0.4)

Table 4.10. The average BMWP scores, with corresponding NTaxa and APST scores for each status category for the sample periods Spring and Summer 2016, with standard deviation in parenthesis.

	Spring 2016			Summer 2016		
	BMWP	**Ntaxa**	**ASPT (BMWP)**	**BMWP**	**Ntaxa**	**ASPT (BMWP)**
All	106.9 (23.3)	16.9 (3.0)	6.3 (0.5)	88.8 (20.2)	15 (3.0)	5.9 (0.5)
Gained	107.7 (24.3)	16.8 (3.4)	6.4 (0.4)	94.3 (20.2)	15.6 (3.2)	6 (0.3)
Lost	101.6 (24.5)	16.8 (3.2)	6 (0.5)	79.1 (20.2)	13.9 (2.9)	5.6 (0.5)
Maintained	110.7 (20.2)	17.2 (2.3)	6.4 (0.5)	91.4 (17.5)	15.3 (2.7)	6 (0.5)

Table 4.11. The average BMWP scores, with corresponding NTaxa and APST scores for each status category for the sample periods Spring and Summer 2017, with standard deviation in parenthesis.

	Spring 2017			Summer 2017		
	BMWP	**Ntaxa**	**ASPT (BMWP)**	**BMWP**	**Ntaxa**	**ASPT (BMWP)**
All	128.4 (27.9)	20.1 (3.9)	6.3 (0.4)	112.6 (28.7)	18.6 (4.3)	6 (0.4)
Gained	134.2 (26.8)	21.1 (3.7)	6.3 (0.4)	115.2 (37.0)	19.1 (5.6)	6 (0.6)
Lost	120.2 (31.9)	19.1 (4.7)	6.3 (0.4)	108.9 (25.8)	18.3 (4.2)	5.9 (0.2)
Maintained	130.1 (23.3)	20.2 (2.9)	6.4 (0.4)	113.6 (21.2)	18.5 (2.8)	6.1 (0.4)

4.3.3. Historical PSI scores

The PSI scores generated for the sampling periods 2007, 2008 and 2009 (labelled 2009A) and for the sampling periods 2010, 2011 and 2012 (labelled 2012A) from EPA monitoring data for 286 of the high status sites recorded through-out Ireland, indicate that the majority of sites were either minimally sedimented/unsedimented (i.e. PSI scores between 81-100) or slightly sedimented (PSI scores 61-80) (Table 4.12). Of the 286 sites, only two sites in 2009A and one site in 2012A were moderately sedimented. A significant difference between all of the 2009A and 2012A PSI scores was found (Wilcoxon paired test, $p<0.01$), with the scores in 2009A being less than 2012A ($p<0.01$). Of the 148 sites that continuously maintained a high status rating (e.g. Q-value of 4.5 or 5) over the 2009A and 2012A periods, no significant difference between the PSI scores of 2009A and 2012A were found (Wilcoxon paired test). Eighty-three sites improved from below high to high status (e.g. from Q-value 3, 3.5 or 4 to 4.5) and within these sites a significant difference in PSI scores between 2009A and 2012A was found (Wilcoxon paired test, $p<0.01$), with 2009A scores being less than 2012A ($p < 0.01$). Fifty-five sites deteriorated from high to below high status between 2009A to 2012A, but no significant difference in PSI scores between 2009A and 2012A for these sites was found. Within the minimally sedimented/unsedimented category, there was a net increase of twelve sites that had Gained in status (i.e. went from a Q-value of below 4.5 to 4.5 or 5), and nine sites that Maintained status, between 2009A and 2012A. However, there, was a net loss of two sites that Lost status from the minimally sedimented/unsedimented category, while one Maintained site became moderately sedimented.

Table 4.12. Number of sites with PSI scores as per Extence et al. (2013) of: 81-100 - Minimally sedimented/unsedimented; 61-80 - Slightly sedimented; 41-60 Moderately sedimented; 21-40 Sedimented; and 0-20 Heavily sedimented; for each status category (Gained, Lost and Maintained) based on EPA historical data-set for 2009A (2007, 2008 and 2009) and 2012A (2010, 2011 and 2012).

Interpretation of PSI scores	**2009A**				**2012A**			
PSI River bed condition	**Gained**	**Lost**	**Maint.**	**Total**	**Gained**	**Lost**	**Maint.**	**Total**
81–100 Min. sedim./unsedim.	66	43	124	233	78	41	133	252
61–80 Slightly sedimented	16	11	24	51	6	14	14	34
41–60 Moderately sedimented	1	1		2			1	1
21–40 Sedimented								
0–20 Heavily sedimented								

4.3.4. Physical sediment properties

The average Scope, Depth, Tile, % Fine and Quorer scores are presented in Tables 4.13 and 4.14, with a full list of scores for the sixty-five sample sites being presented in Appendix D, Tables D8 and D9. The highest % Fine score occurred in Summer 2017 at site 32O040250 (64 %), with the highest Quorer score (6.3 g/m^2) occurring in Summer 2016 at site 34Y020275. The highest Scope score occurred at site 26I030300 in Spring 2016 and the highest Depth score (14.67 cm) occurred at site 34Y020275 in Spring 2016. For all the physical sediment variables, for each sampling period, the only significant difference between any of the status categories, was in Summer 2017 for the Quorer between Gained and Lost, with a p values of 0.03. Significant differences (yearly) for Depth and Tile scores recorded in Spring 2016 and those recorded in Spring 2017 were found, both at $p<0.01$, while Scope ($p = 0.011$) and Quorer ($p = 0.013$) scores recorded in Summer 2016 were significantly different from those recorded in Summer 2017.

Table 4.13. The average Scope (%), Depth (cm) and Tile (score between 1-5) scores for the sixty-five sample sites during Spring 2016, Summer 2016, Spring 2017 and Summer 2017, with standard deviations in parenthesis.

	Scope (%)				Depth (cm)				Tile (score bet. 1-5)			
	Spring 2016	Summer 2016	Spring 2017	Summer 2017	Spring 2016	Summer 2016	Spring 2017	Summer 2017	Spring 2016	Summer 2016	Spring 2017	Summer 2017
All	15.2 (15.8)	14 (18.5)	14 (18.6)	8.5 (14.6)	1.6 (2.4)	0.6 (1.1)	0.8 (1.8)	0.5 (2)	2.6 (1)	3 (1.1)	3 (1)	3 (1)
Gained	15.5 (17.5)	12.4 (19.4)	16.1 (23.3)	7.6 (13.5)	2.3 (3.3)	0.9 (1.5)	1 (2.8)	1 (3.1)	2.8 (1.1)	3.1 (1.1)	3.2 (1)	3.4 (1.1)
Lost	15.7 (15.5)	12.1 (20.7)	12.9 (11.9)	8.8 (18.4)	1 (1.9)	0.4 (1.1)	1 (1)	0.4 (1.2)	2.5 (1.1)	3.1 (0.9)	2.8 (1.1)	3 (0.9)
Maintained	14.5 (11.4)	17 (15.1)	13 (17)	9.1 (12)	1.4 (1.3)	0.6 (1)	0.4 (0.5)	0.4 (1)	2.5 (0.8)	2.9 (1.1)	3 (0.9)	2.6 (0.6)

Table 4.14. The average % Fine (%) and Quorer (g/m^2) scores for the sixty-five sample sites during Spring 2016, Summer 2016, Spring 2017 and Summer 2017, with standard deviations in parenthesis.

	% Fine				Quorer (g/m^2)			
	Spring 2016	Summer 2016	Spring 2017	Summer 2017	Spring 2016	Summer 2016	Spring 2017	Summer 2017
All	11.3 (12.6)	10.4 (12)	11.9 (13.6)	9.9 (13.5)	0.6 (1.1)	0.3 (0.4)	0.5 (0.5)	0.2 (0.2)
Gained	10.5 (14.5)	8 (9)	12.6 (17.7)	10.8 (16.6)	0.9 (1.3)	0.3 (0.3)	0.6 (0.6)	0.2 (0.1)
Lost	12.7 (13.2)	11.4 (15.4)	10.5 (8.1)	11.4 (14.8)	0.5 (1.1)	0.4 (0.6)	0.4 (0.4)	0.2 (0.1)
Maintained	10.6 (7.1)	11.8 (10.8)	12.5 (13.2)	7.5 (6.3)	0.5 (0.2)	0.4 (0.3)	0.4 (0.2)	0.2 (0.3)

Seasonal differences were found for Depth, Tile and Quroer scores between Spring 2016 and Summer 2016, all at p <0.01, and for Scope (p< 0.01), Depth (p=0.043), Sus (p = 0.032) and Quorer (p< 0.01) between Spring 2017 and Summer 2017. Spring 2016 scores for Depth were greater, than Summer 2016 and Spring 2017 scores, although for Tile, Spring 2016 scores were less than Summer 2016 and Spring 2017. Where relevant, Spring 2016, Summer 2016 and Spring 2017 scores were greater than Summer 2017 for Scope, Depth, % Fine and Quorer.

Within the Gained status category, there was a significant difference between sites for: Depth (p <0.01) and Tile (p =0.014) values recorded in Spring 2016 and those recorded in Spring 2017; Depth (p = 0.037), Scope (p<0.01) and Quorer (p<0.01) values recorded in Spring 2016 and those recorded in Summer 2016; and Scope (p<0.01), Depth (p = 0.058), and Quorer (p<0.01) values recorded in Spring 2017 and those recorded in Summer 2017. Within the Lost status category, there was a significant difference between sites for: Depth (p=0.049) values recorded in Spring 2016 and those recorded in Summer 2016; Tile (p =0.048) values recorded in Spring 2016 and those recorded in Spring 2017; and Scope (p=0.014) and Quorer (p = 0.011) values recorded in Spring 2017 and those recorded in Summer 2017. Within the Maintained status category, there was a significant difference between sites for: Depth (p<0.01) and Tile (p = 0.015) values recorded in Spring 2016 and those recorded in Spring 2017; Scope (p = 0.05) values recorded in Summer 2016 and those recorded in Summer 2017; Depth (p=0.033) values recorded in Spring 2016 and those recorded in Summer 2016; and % Fine (p = 0.036) and Quorer (p<0.01) values recorded in Spring 2017 and those recorded in Summer 2017. Additionally for Maintained sites, p values close to significance were found for % Fine (p=0.063) for Summer 2016 against Summer 2017; and for Tile (p =0.055) for Spring 2016 against Summer 2016.

4.3.5. Physico-chemical properties

pH values in Spring 2017 ranged from a low of 6.82 at the site 25B150050 to a high of 8.19 at site 34C030150. Conductivity values in Spring 2016 ranged from a low of 58 µS/cm at site 31R010100 to a high of 571 µS/cm at site 26D070700, from a low of 36.9 µS/cm at site 26Y010200 to a high of 762.3 µS/cm at site 25D100200 in Summer 2016, from a low of 62.2 µS/cm at site 26I030300 to a high of 646 µS/cm at site 26D070700 in Spring 2017, and from a low of 35.9 µS/cm at site 31R010100 to a high of 606 µS/cm at site 26D070700 in Summer 2017. The average dissolved oxygen (DO) reading in Summer 2016, was 98.4 % saturation, with the lowest recording being 79.4 %. In Spring 2017 the average DO reading was 116.7 % saturation, with the lowest recording being 87.8 %, and in Summer 2017 the average DO reading was 124.2 % saturation, with the lowest recording being 82.8 %.

The EPA Parameters of Water Quality (2001) assigns a Q-value rating of 5 (high status) to rivers with a SRP (MRP) value of 0.015 mg/l P or less, a Q-value of 4.5 (high status) to SRP values of between 0.02 mg/l P and 0.015 mg/l P, and a Q-value of 4 (Good status) to SRP values of between 0.03 mg/l P and 0.02 mg/l P. In Spring 2016 two sites, one Gained and one Lost, had a SRP value greater than 0.015mg/l P, with the highest value being 0.0206 mg/l P (Lost site). In Summer 2016, eleven sites had a SRP value greater than 0.015 mg/l P. Of these, five sites (two Lost sites and three Gained sites) had a SRP value greater than 0.02 mg/l P, with the highest value being 0.104 mg/L at the Lost site 26I030400 (although this was likely a contaminated sample

– see EPA measurement below). No sites in Spring 2017 had a SRP value greater than 0.015 mg/l P. In Summer 2017, one site (Lost - SRP value of 0.028 mg/l P) had a SRP value of greater than 0.015 mg/l P.

Based on the EPA laboratory analysis from Summer 2016, ten sites had o-phosphate values greater than 0.015 mg/l P, of which seven sites (one Gained, three Maintained and two Lost sites) had values greater than 0.02 mg/l P, with the highest reading being 0.04 mg/l P at the Maintained site 35C030200. The EPA laboratory analysis recorded a value of less than 0.01 mg/L P for site 26I030400. The EPA laboratory analysis from Summer 2016 found nine sites had ammonia concentrations greater than 0.02 mg/l N, with the highest value being 0.033 mg/l N. Nineteen sites had TON and Nitrate concentrations greater than 0.2 mg/l N, with values at these nineteen sites ranging from 0.23 mg/l N to 0.86 mg/l N, for both TON and Nitrate. All sites had Nitrite values less than or equal to 0.004 mg/l N. Thirteen sites had a BOD value greater than 1 mg/l O2, of which two sites (one Lost – 2.5 mg/l O2; one Maintained - 2.5 mg/l O2) had values in excess of 1.65 mg/l O2, although a value of $\leq$ 5 mg/l O2 is acceptable for Salmonid waters (EPA, 2001).

4.3.6. Spearman rank correlations between physical and biological variables

Spearman rank correlations between each of the physical sediment variables (Scope, Depth, Tile, % Fine and Quorer) and biological indices (PSI, CoFSI, BMWP, N-taxa (BMWP), ASPT (BMWP), WHPT, N-taxa (WHPT) and ASPT (WHPT)), for each sampling period, Spring 2016, Summer 2016, Spring 2017 and Summer 2017 are presented in Tables 4.15 a and b and 4.16 a and b, respectively. In general, each of the physical sediment variables across all seasons displayed significant moderate to strong relationships with each of the other physical sediment variables. The strongest relationships were observed between Quorer and Tile (excluding Summer 2017), and between Scope and % Fine. The weakest relationship occurred in Spring 2016 between Depth and Quorer. Of the biological variables, E-PSI, PSI and ASPT (primarily for WHPT) had the strongest significant relationships with the physical sediment variables. Negative weak to moderate relationships between these variables and the physical variables were observed, with stronger relationships tending to occur in the Summer sampling periods. The strongest relationship occurring in Spring 2016 was between E-PSI and % Fine; in Summer 2016 was between E-PSI and Scope; in Spring 2017 was between PSI and Quorer; and in Summer 2017 was between PSI and Tile. While LIFE scores did show some relationship with the physical variables, Scope, Depth, Tile and % Fine, no relationship was observed with Quorer. Strong/very strong relationships were observed between each of the E-PSI, PSI, LIFE and ASPT (WHPT) biological variables. With the exception of Depth in Summer 2016, no significant relationship between CoFSI and any of the physical sediment variables was observed, nor between CoFSI and PSI, E-PSI or LIFE. A comparison of each sediment variable across each season is presented in Table 4.17.

Table 4.15. Spearman rank correlations between each physical sediment variable (Scope, Depth, Tile, % Fine and Quorer) and biological indices (PSI, CoFSI, BMWP, N-taxa (BMWP), ASPT (BMWP), WHPT, N-taxa (WHPT) and ASPT (WHPT)), for a) Spring 2016 and b) Summer 2016.

a) **. Correlation is significant at the 0.01 level (2-tailed). *. Correlation is significant at the 0.05 level (2-tailed).

Spring 2016	Scope	Depth	Tile	% Fine	Quorer	PSI	CoFSI	E-PSI	LIFE	BMWP	N (BMWP)	ASPT (BMWP)	WHPT	N (WHPT)	ASPT (WHPT)
Scope	1	0.582**	0.539**	0.755**	0.434**	-0.347**	-0.007	-0.554**	-0.405**	0.117	0.165	-0.086	-0.007	0.092	-0.360**
Depth	0.582**	1	0.462**	0.486**	0.388**	-0.342*	0.053	-0.455**	-0.357**	-0.007	0.064	-0.153	0.006	0.086	-0.300*
Tile	0.539**	0.462**	1	0.530**	0.734**	-0.293*	0.008	-0.494**	-0.280*	0.057	0.128	-0.139	-0.094	-0.021	-0.316*
% Fine	0.755**	0.486**	0.530**	1	0.451**	-0.437**	-0.133	-0.595**	-0.506**	-0.097	-0.021	-0.252*	-0.174	-0.038	-0.495**
Quorer	0.434**	0.388**	0.734**	0.451**	1	-0.211	-0.051	-0.381**	-0.232	-0.013	0.026	-0.100	-0.132	-0.119	-0.198
PSI	-0.347**	-0.342*	-0.293*	-0.437**	-0.211	1	0.160	0.806**	0.798**	0.311*	0.054	0.760**	0.257*	0.007	0.814**
CoFSI	-0.007	0.053	0.008	-0.133	-0.051	0.160	1	0.047	0.132	0.702**	0.715**	0.429**	0.838**	0.824**	0.364**
E-PSI	-0.554**	-0.455**	-0.494**	-0.595**	-0.381**	0.806**	0.047	1	0.717**	0.070	-0.151	0.517**	0.090	-0.136	0.688**
LIFE	-0.405**	-0.357**	-0.280*	-0.506**	-0.232	0.798**	0.132	0.717**	1	0.195	-0.004	0.588**	0.215	-0.007	0.779**
BMWP	0.117	-0.007	0.057	-0.097	-0.013	0.311*	0.702**	0.070	0.195	1	0.945**	0.724**	0.884**	0.814**	0.517**
N (BMWP)	0.165	0.064	0.128	-0.021	0.026	0.054	0.715**	-0.151	-0.004	0.945**	1	0.477**	0.873**	0.885**	0.292*
ASPT (BMWP)	-0.086	-0.153	-0.139	-0.252*	-0.100	0.760**	0.429**	0.517**	0.588**	0.724**	0.477**	1	0.586**	0.370**	0.821**
WHPT	-0.007	0.006	-0.094	-0.174	-0.132	0.257*	0.838**	0.090	0.215	0.884**	0.873**	0.586**	1	0.946**	0.499**
N (WHPT)	0.092	0.086	-0.021	-0.038	-0.119	0.007	0.824**	-0.136	-0.007	0.814**	0.885**	0.370**	0.946**	1	0.229
ASPT (WHPT)	-0.360**	-0.300*	-0.316*	-0.495**	-0.198	0.814**	0.364**	0.688**	0.779**	0.517**	0.292*	0.821**	0.499**	0.229	1

b) **. Correlation is significant at the 0.01 level (2-tailed). *. Correlation is significant at the 0.05 level (2-tailed).

Summer 2016	Scope	Depth	Tile	% Fine	Quorer	PSI	CoFSI	E-PSI	LIFE	BMWP	N (BMWP)	ASPT (BMWP)	WHPT	N (WHPT)	ASPT (WHPT)
Scope	1	0.508**	0.562**	0.689**	0.531**	-0.588**	-0.109	-0.618**	-0.571**	-0.049	0.137	-0.388**	-.0164	0.019	-0.553**
Depth	0.508**	1	0.605**	0.532**	0.608**	-0.348**	-0.317*	-0.400**	-0.267*	-0.251	-0.106	-0.363**	-0.311*	-0.164	-0.407**
Tile	0.562**	0.605**	1	0.405**	0.763**	-0.571**	0.070	-0.520**	-0.377**	-0.041	0.191	-0.455**	0.001	0.165	-0.388**
% Fine	0.689**	0.532**	0.405**	1	0.337**	-0.2254	-0.155	-0.271*	-0.355**	-0.054	0.036	-0.143	-0.157	-0.059	-0.266*
Quorer	0.531**	0.608**	0.763**	0.337**	1	-0.472**	0.114	-0.532**	-0.169	0.044	0.223	-0.327*	0.058	0.227	-0.272*
PSI	-0.588**	-0.348**	-0.571**	-0.225	-0.472**	1	-0.063	0.868**	0.736**	0.049	-0.197	0.571**	0.070	-0.153	0.703**
CoFSI	-0.109	-0.317*	0.070	-0.155	0.114	-0.063	1	-0.176	0.210	0.650**	0.681**	0.237	0.826**	0.835**	0.349**
E-PSI	-0.618**	-0.400**	-0.520**	-0.271*	-0.532**	0.868**	-0.176	1	0.589**	-0.084	-0.307*	0.414**	-0.075	-0.287*	0.550**
LIFE	-0.571**	-0.267*	-0.377**	-.0355**	-0.169	0.736**	0.210	0.589**	1	0.178	-0.025	0.531**	0.268*	0.056	0.752**
BMWP	-0.049	-0.251	-0.041	-0.054	0.044	0.049	0.650**	-0.084	0.178	1	0.918**	0.604**	0.871**	0.796**	0.530**
N (BMWP)	0.137	-0.106	0.191	0.036	0.223	-0.197	0.681**	-0.307*	-0.025	0.918**	1	0.302*	0.806**	0.846**	0.281*
ASPT (BMWP)	-0.388**	-0.363**	-0.455**	-0.143	-0.327*	0.571**	0.237	0.414**	.0531**	0.604**	0.302*	1	0.486**	0.277*	0.798**
WHPT	-0.164	-0.311*	0.001	-0.157	0.058	0.070	0.826**	-0.075	0.268*	0.871**	.0806**	0.486**	1	0.934**	0.545**
N (WHPT)	0.019	-0.164	0.165	-0.059	0.227	-0.153	0.835**	-0.287*	0.056	0.796**	0.846**	0.277*	0.934**	1	0.276*
ASPT (WHPT)	-0.553**	-0.407**	-0.388**	-0.266*	-0.272*	0.703**	0.349**	0.550**	0.752**	0.530**	0.281*	0.798**	0.545**	0.276*	1

Table 4.16. Spearman rank correlations between each physical sediment variable (Scope, Depth, Tile, % Fine and Quorer) and biological indices (PSI, CoFSI, BMWP, N-taxa (BMWP), ASPT (BMWP), WHPT, N-taxa (WHPT) and ASPT (WHPT)), for a) Spring 2017 and b) Summer 2017.

a) **. Correlation is significant at the 0.01 level (2-tailed). *. Correlation is significant at the 0.05 level (2-tailed).

Spring 2017	**Scope**	**Depth**	**Tile**	**% Fine**	**Quorer**	**PSI**	**CoFSI**	**E-PSI**	**LIFE**	**BMWP**	**N (BMWP)**	**ASPT (BMWP)**	**WHPT**	**N (WHPT)**	**ASPT (WHPT)**
Scope	1	0.610**	0.560**	0.665**	0.489**	-0.249*	-0.106	-0.352**	-0.214	-0.116	-0.100	-0.104	-0.213	-0.127	-0.337**
Depth	0.610**	1	0.466**	0.519**	0.519**	-0.241	-0.207	-0.373**	-0.245*	-0.205	-0.212	-0.105	-0.308*	-0.245*	-0.377**
Tile	0.560**	0.466**	1	0.545**	0.693**	-0.272*	-0.108	-0.358**	-0.058	-0.141	-0.134	-0.085	-0.199	-0.152	-0.286*
% Fine	0.665**	0.519**	0.545**	1	0.469**	-0.240	-0.180	-0.383**	-0.240	-0.169	-0.189	-0.058	-0.274*	-0.219	-0.311*
Quorer	0.489**	0.519**	0.693**	0.469**	1	-0.426**	-0.198	-0.406**	-0.204	-0.276*	-0.224	-0.303*	-0.305*	-0.246*	-0.426**
PSI	-0.249*	-0.241	-0.272*	-0.240	-0.426**	1	0.086	0.824**	0.718**	0.072	-0.054	0.465**	0.204	0.032	0.633**
CoFSI	-0.106	-0.207	-0.108	-0.180	-0.198	0.086	1	-0.020	0.193	0.866**	0.870**	0.397**	0.859**	0.856**	0.373**
E-PSI	-0.352**	-0.373**	-0.358**	-0.383**	-0.406**	0.824**	-0.020	1	0.694**	-0.056	-0.185	0.387**	0.058	-0.113	0.617**
LIFE	-0.214	-0.245*	-0.058	-0.240	-0.204	0.718**	0.193	0.694**	1	0.150	0.018	0.540**	0.236	0.052	0.727**
BMWP	-0.116	-0.205	-0.141	-0.169	-0.276*	0.072	0.866**	-0.056	0.150	1	0.962**	0.565**	0.941**	0.935**	0.459**
N (BMWP)	-0.100	-0.212	-0.134	-0.189	-0.224	-0.054	0.870**	-0.185	0.018	0.962**	1	0.339**	0.923**	0.970**	0.290*
ASPT (BMWP)	-0.104	-0.105	-0.085	-0.058	-0.303*	0.465**	0.397**	0.387**	0.540**	0.565**	0.339**	1	0.513**	0.340**	0.793**
WHPT	-0.213	-0.308*	-0.199	-0.274*	-0.305*	0.204	0.859**	0.058	0.236	0.941**	0.923**	0.513**	1	0.954**	0.555**
N (WHPT)	-0.127	-0.245*	-0.152	-0.219	-0.246*	0.032	0.856**	-0.113	0.052	0.935**	0.970**	0.340**	0.954**	1	0.322**
ASPT (WHPT)	-0.337**	-0.377**	-0.286*	-0.311*	-0.426**	0.633**	0.373**	0.617**	0.727**	0.459**	0.290*	0.793**	0.555**	0.322**	1

b) **. Correlation is significant at the 0.01 level (2-tailed). *. Correlation is significant at the 0.05 level (2-tailed).

Summer 2017	**Scope**	**Depth**	**Tile**	**% Fine**	**Quorer**	**PSI**	**CoFSI**	**E-PSI**	**LIFE**	**BMWP**	**N (BMWP)**	**ASPT (BMWP)**	**WHPT**	**N (WHPT)**	**ASPT (WHPT)**
Scope	1	0.638**	0.763**	0.805**	0.611**	-0.496**	-0.026	-0.490**	-0.234	-0.018	0.039	-0.203	0.023	0.121	-0.292*
Depth	0.638**	1	0.586**	0.630**	0.472**	-0.411**	0.090	-0.421**	-0.264*	0.168	0.177	-0.126	0.157	0.239	-0.183
Tile	0.763**	0.586**	1	0.613**	0.589**	-0.557**	0.007	-0.528**	-0.366**	0.033	0.094	-0.302*	0.012	0.135	-0.379**
% Fine	0.805**	0.630**	0.613**	1	0.475**	-0.508**	-0.092	-0.530**	-0.296*	0.037	0.074	-0.052	0.031	0.115	-0.275*
Quorer	0.611**	0.472**	0.589**	0.475**	1	-0.351**	0.132	-0.443**	-0.201	-0.033	0.060	-0.407**	-0.004	0.120	-0.366**
PSI	-0.496**	-0.411**	-0.557**	-0.508**	-0.351**	1	-0.090	0.837**	0.779**	-0.123	-0.235	0.398**	-0.102	-0.285*	0.657**
CoFSI	-0.026	0.090	0.007	-0.092	0.132	-0.090	1	-0.200	0.086	0.758**	0.794**	0.159	0.779**	0.802**	0.175
E-PSI	-0.490**	-0.421**	-0.528**	-0.530**	-0.443**	0.837**	-0.200	1	0.681**	-0.256*	-0.369**	0.274*	-0.209	-0.393**	0.579**
LIFE	-0.234	-0.264*	-0.366**	-0.296*	-0.201	0.779**	0.086	0.681**	1	0.055	-0.051	0.335**	0.137	-0.059	0.783**
BMWP	-0.018	0.168	0.033	0.037	-0.033	-0.123	0.758**	-0.256*	0.055	1	0.970**	0.458**	0.954**	0.925**	0.356**
N (BMWP)	0.039	0.177	0.094	0.074	0.060	-0.235	0.794**	-0.369**	-0.051	0.970**	1	0.273*	0.934**	0.960**	0.202
ASPT (BMWP)	-0.203	-0.126	-0.302*	-0.052	-0.407**	0.398**	0.159	0.274*	0.335**	0.458**	0.273*	1	0.421**	0.238	0.696**
WHPT	0.023	0.157	0.012	0.031	-0.004	-0.102	0.779**	-0.209	0.137	0.954**	0.934**	0.421**	1	0.956**	0.395**
N (WHPT)	0.121	0.239	0.135	0.115	0.120	-0.285*	0.802**	-0.393**	-0.059	0.925**	0.960**	0.238	0.956**	1	0.161
ASPT (WHPT)	-0.292*	-0.183	-0.379**	-0.275*	-0.366**	0.657**	0.175	0.579**	0.783**	0.356**	0.202	0.696**	0.395**	0.161	1

Table 4.17. Spearman rank correlations between each physical sediment variables (Scope, Depth, Tile, % Fine and Quorer) and each sample season.

		Scope				**Depth**				**Tile**				**% Fine**				**Quorer**			
		Spring 16	**Summer 16**	**Spring 17**	**Summer 17**	**Spring 16**	**Summer 16**	**Spring 17**	**Summer 17**	**Spring 16**	**Summer 16**	**Spring 17**	**Summer 17**	**Spring 16**	**Summer 16**	**Spring 17**	**Summer 17**	**Spring 16**	**Summer 16**	**Spring 17**	**Summer 17**
Scope	**Spring 16**	1	.619**	.571**	.621**	.582**	.628**	.422**	.383**	.539**	.527**	.389**	.562**	.755**	.447**	.684**	.581**	.434**	.298*	.485**	.295*
	Summer 16	.619**	1	.636**	.492**	.248	.508**	.433**	.449**	.536**	.562**	.514**	.365*	.641**	.689**	.668**	.481**	.370**	.531**	.603**	.267*
	Spring 17	.571**	.636**	1	.762**	.327*	.469**	.610**	.575**	.441**	.612**	.560**	.687**	.636**	.480**	.665**	.598**	.398**	.463**	.489**	.419**
	Summer 17	.621**	.492**	.762**	1	.405**	.556**	.662**	.638**	.503**	.452**	.511**	.763**	.681**	.447**	.568**	.805**	.316*	.444**	.435**	.611**
Depth	**Spring 16**	.582**	.248	.327*	.405**	1	.581**	.414**	.330*	.462**	.429**	.303*	.397**	.486**	.126	.400**	.420**	.388**	.148	.253	.191
	Summer 16	.628**	.508**	.469**	.555**	.581**	1	.514**	.283*	.614**	.605**	.370**	.566**	.514**	.532**	.463**	.598**	.498**	.608**	.523**	.298*
	Spring 17	.422**	.433**	.610**	.662**	.414**	.514**	1	.561**	.364**	.538**	.466**	.588**	.621**	.345**	.519**	.694**	.354**	.386**	.519**	.319*
	Summer 17	.383**	.449**	.575**	.638**	.330*	.283*	.561**	1	.337*	.303*	.372**	.586**	.525**	.382**	.447**	.630**	.338*	.322*	.321*	.472**
Tile	**Spring 16**	.539**	.536**	.441**	.503**	.462**	.614**	.364**	.337*	1	.591**	.544**	.567**	.530**	.287*	.533**	.407**	.734**	.474**	.503**	.281*
	Summer 16	.527**	.562**	.612**	.452**	.429**	.605**	.538**	.303*	.591**	1	.562**	.642**	.573**	.405**	.510**	.326*	.606**	.763**	.576**	.230
	Spring 17	.389**	.514**	.560**	.511**	.303*	.370**	.466**	.372**	.544**	.562**	1	.499**	.497**	.331**	.545**	.380**	.496**	.513**	.693**	.408**
	Summer 17	.562**	.365*	.687**	.763**	.397**	.566**	.588**	.586**	.567**	.642**	.499**	1	.578**	.262	.418**	.613**	.446**	.500**	.406**	.589**
% Fine	**Spring 16**	.755**	.641**	.636**	.681**	.486**	.514**	.621**	.525**	.530**	.573**	.497**	.578**	1	.489**	.761**	.631**	.451**	.383**	.543**	.366**
	Summer 16	.447**	.689**	.480**	.447**	.126	.532**	.345**	.382**	.287*	.405**	.331**	.262	.489**	1	.469**	.478**	.240	.406**	.459**	.127
	Spring 17	.684**	.668**	.665**	.568**	.400**	.463**	.519**	.447**	.533**	.510**	.545**	.418**	.761**	.469**	1	.527**	.378**	.380**	.469**	.271*
	Summer 17	.581**	.481**	.598**	.805**	.420**	.598**	.694**	.630**	.407**	.326*	.380**	.613**	.631**	.478**	.527**	1	.358**	.345**	.543**	.524**
Quorer	**Spring 16**	.434**	.370**	.398**	.316*	.388**	.498**	.354**	.338*	.734**	.606**	.496**	.446**	.451**	.240	.378**	.358**	1	.500**	.348**	.285*
	Summer 16	.298*	.531**	.463**	.444**	.148	.608**	.386**	.322*	.474**	.763**	.513**	.500**	.383**	.406**	.380**	.345**	.500**	1	.506**	.454**
	Spring 17	.485**	.603**	.489**	.435**	.253	.523**	.519**	.321*	.503**	.576**	.693**	.406**	.543**	.459**	.469**	.543**	.348**	.506**	1	.339**
	Summer 17	.295*	.267*	.419**	.611**	.191	.298*	.319*	.472**	.281*	.230	.408**	.589**	.366**	.127	.271*	.524**	.285*	.454**	.339**	1

**. Correlation is significant at the 0.01 level (2-tailed). *. Correlation is significant at the 0.05 level (2-tailed).

4.4. Discussion

Sediment pressures, particularly associated with agriculture and forestry, have been cited as a potential factor contributing to declines in the number of HSWs (RBMP, 2018; White et al., 2014). To date this has received little attention, and although some studies assessing sediment as a pressure, e.g. Conroy et al, 2016b, have included high status sites as part of their site selection, the number included tends to be limited to perhaps one or two sites. Here the impacts of sediment on sixty-five high status river sites that were determined to have either: "Lost" their high status (e.g. gone from high to good, moderate, poor or bad); consistently "Maintained" their high status; or "Gained" in status (e.g. from good to high) was assessed.

The general trend across all sample sites and seasons was for invertebrate taxa that are either highly sensitive or moderately sensitive to sedimentation to dominate in terms of taxa present and abundances. This was reflected in the PSI scores which, with the exception of four sites across all sampling periods, were all above the slightly–sedimented base score of sixty-one. Similarly, E-PSI scores, which were predominantly above 70%, indicated a dominance of sediment sensitive taxa. However, significant PSI and E-PSI score differences were found between sites classified as Lost and Maintained for all sampling periods, with Maintained sites scoring higher than Lost sites, indicating that invertebrate communities in Maintained sites were more sediment sensitive. Additionally, Lost sites had a greater number and proportion of sites classified as slightly sedimented, in comparison to Maintained and Gained. While the significant differences between Lost and Maintained for PSI and E-PSI scores implies that deterioration in status is associated with sedimentation, the lack of any difference in PSI and E-PSI scores between Lost and Gained highlights an important caveat. The dominance of taxa highly sensitive or moderately sensitive to sedimentation was again observed in the historical EPA data-set. However, while there was a significant difference in the PSI scores between the EPA data-set sites that Gained in status between 2009A and 2012A, no difference was observed for sites that deteriorated (Lost) in status.

Although only recently introduced, several studies have utilised PSI scores for assessing sediment pressures (Poole et al., 2013; Glendell et al., 2014; Conroy et al., 2016a; Bradley et al., 2017; Extence et al., 2017). Extence et al. (2017) for example, found using a national data set, a significantly strong (r^2 = 59.7%) relationship between PSI scores and a channel substrate index (CSI) designed to assess levels of fine sediment. Glendell et al. (2014) similarly, found a significant relationship between PSI and % fine bed sediment cover, although no relationship between PSI and three other sediment assessment variables (two suspended sediment – including a Quorer method; and % exceedance method) was observed. Conroy et al. (2016a) and Turley et al. (2014) both found PSI to correlate with sediment cover, although Conroy et al. found a stronger relationship with sediment cover for % EPT (Ephemeroptera, Plecoptera and Trichoptera).

Here the Spearman rank analysis found both PSI and E-PSI were more associated with with the physical sediment variables, although only negative weak to moderate relationships were observed. Glendell et al. (2014) suggests the lack of a relationship between PSI and the suspended sediment variables in their study may have been related to the sample resolution, with suspended sediment being measured at the patch scale, while invertebrate monitoring for the PSI was conducted at the reach level.

While similar sampling practices (i.e. patch for suspended sediment and reach scale for invertebrates) were conducted in this study, with the exception of PSI in Spring 2016 and Summer 2017, a moderate relationship was observed between PSI, E-PSI and suspended sediment (Quorer).

In contrast to the PSI and E-PSI scores, CoFSI only found a significant difference between Lost and Maintained in one sampling period (Spring 2016). Other studies found a strong negative correlation between CoFSI and fine sediment levels, and strong positive correlations between CoFSI and PSI, E-PSI and LIFE metrics (Murphy et al., 2015; Turley et al., 2016). Here, however, Spearman rank correlations found no relationship between CoFSI scores and (with the exception of Depth in Summer 2016) any of the physical sediment analysis methods, in strong contrast to the PSI and E-PSI metrics. This perhaps suggests that the CoFSI metric may need to be re-appraised prior to application in an Irish context, especially in relation to minimally impacted sites.

Despite the observed relationships between PSI, E-PSI and the sediment variables, with the exception of Gained against Lost for the Quorer method in Summer 2016 and Summer 2017, no difference between the three status categories was observed for any of the five physical sedimentation analysis methods. Contrary to the PSI and E-PSI scores, the lack of a significant difference between Lost and Maintained, and Lost and Gained, for the physical sediment variables implies sedimentation is not a factor associated with the deterioration of the HSWs. This contradiction is difficult to explain. While sampling resolution reasons may hold true for the two re-suspension techniques (Quorer and Tile), the visual assessment method is more of a reach scale assessment. Several studies have demonstrated the benefits of visual assessment methods for assessing sedimentation (Sutherland et al., 2012; Zweig and Rabeni, 2001; Conroy et al., 2016c). While visual assessments may be somewhat subjective in nature and potentially susceptible to operator bias (but see Conroy et al., 2016c), this may be limited when, as in this study, all assessments are carried out by a single operator (Zweig and Rabeni, 2001).

On the other hand, Conroy et al. (2016a) in a mescom study, found PSI scores at very high sediment loadings, to be far in excess of that expected, and questioned the suitability of the PSI metric to accurately assess sedimentation pressures. Additionally, Buendia et al. (2013) found *Baetis* to be sediment tolerant, which differs from the sensitive classification within the PSI metric (Extence et al., 2013), although other studies have reported declines in the abundance of *Baetis rhodani* in response to increased levels of sediment (Larsen et al., 2011). Resilience of taxa, conferred from for example, less specialised feeding habits and high fecundity rates, may potentially lead to mis-leading conclusions with biotic metrics (Buendia et al., 2013), such as with the PSI and E-PSI metrics. Additionally, the potential for the chemical/nutrient composition of the sediment to alter the chemical composition of receiving waters (Bilotta and Brazier, 2008), which was not assessed in this study, may impact on PSI scores.

Given that some sedimentation occurs naturally in rivers, ideally sediment metric scores should be compared with expected/reference condition scores for associated sample sites (Turley et al., 2015). For example, in an assessment of UK reference sites Bilotta et al. (2012) found a significant difference in the mean background suspended particulate matter (SPM) levels recorded in sites of varying habitat characteristics (slope, altitude etc.) and suggests that E.U. guideline rates for sediment levels (should not exceed 25 mg/L - Freshwater Fish Directive – OJEC, 2006), require modification to accurately reflect the pressures associated with each habitat "type". Similarly, Relyea et al. (2011) found variations in fine sediment levels were related to stream gradient, stream order and the ecoregion in which the streams occurred.

In Ireland rivers are primarily characterised based on hardness (mg/l $CaCO_3$) and slope resulting in twelve possible habitat/character types (Kelly-Quinn et al., 2005). In this study, a proportional representation of river types coded for hardness and slope, for each status category, (based on their occurrence out of a possible 165 West of Ireland HSWs) were selected. However, this perhaps leaves room for error, given that the same number of habitat types were not present within each status category. Additionally, this study does not compare observed results with those expected (primarily due to a lack of accurate data such as alkalinity readings etc., and an appropriate mechanism such as RICT/RIVPACS as employed by the UK Environment Agency), and this is something that should be considered for any future studies. Low to moderate increases in silt may impact invertebrate communities (Larsen et al., 2011), with this potentially having a disproportionally large impact on HSWs, in comparison to the same silt increase in already degraded water-bodies (White et al., 2014). Comparisons of observed vs expected scores may allow for this small increase to be detected relative to the naturally occurring background sediment variability occurring at each site.

As with the habitat characteristics, there is a risk in assuming that biological communities from different rivers and streams share a uniform response to the same pressure, especially given the potential interaction of multiple stressors, and the possibility of certain taxa developing a resilience to specific stressors (Turley et al., 2016). The interaction of stressors may be antagonistic - whereby two or more stressors are acting on the same species therefore their net effect is less than, for example, the same stressors acting individually on different species; synergistic – where a species is only impacted by a combination of stressors; additive – where different stressors act on different taxa; and reversal – where one stressor reverses the impact of another stressor (Jackson et al., 2016), although see also Gieswein et al. (2017). Disentangling specific stressors, and the impact individual parameters have on benthic organisms is therefore difficult (Rempel et al., 2000; Marzin et al., 2012). Matthaei et al. (2010) in an experiment to assess the multiple stressor effects of sedimentation, water abstraction and nutrient enrichment, found the interaction between reduced flow and sediment addition to have the most impact on biological parameters.

Here a strong relationship was observed between LIFE and PSI, while a moderate to strong relationship was observed between LIFE and E-PSI. This may indicate some interaction between flow and sedimentation is responsible for the invertebrate communities present as found by other studies (Glendell et al., 2014; Turley et al., 2016). However, the relationship between the LIFE index and the sediment variables, especially those of the Quorer, implies that only a weak/little relationship between flow and sedimentation exists in these study sites. Turley et al. (2016) found the relationship between LIFE and E-PSI and LIFE and fine sediment weakened as the stream power group associated with the sample site increased, and this should perhaps be incorporated into any future studies assessing the relationship between flow and sedimentation.

Although the sites in this study are or were HSWs, and are therefore likely to be minimally impacted from pollution stressors, significant differences between Lost and Maintained were observed in three of the four sampling periods for ASPT (WHPT) and two of the four seasons for ASPT (BMWP). This implies that organic pollution may be impacting on some sites, while the strong relationship between ASPT (WHPT) and PSI implies some interaction between organic pollution and sedimentation. Sutherland et al. (2012) suggests that the relationship between a biotic index designed for assessing organic pollution, and sedimentation, in their study, may be related to: fine sediment reducing benthic oxygen levels in a similar fashion to the impacts of organic pollution; the smothering of taxa sensitive to organic pollution; or the chemical/nutrient properties of the sediment. Again, this interaction is worthy of further investigation. In contrast, however, the nutrient values (P and N) recorded at the majority of sample sites indicate little or no evidence of nutrient enrichment, although the one off nutrient assessments as conducted in this study, are not likely to capture potentially large and rapid changes in nutrient emissions (Bowes et al., 2009; Cassidy and Jordan, 2011; Halliday et al., 2012; Bowes et al., 2015). Several studies have demonstrated the benefits of employing high resolution water quality monitoring for nutrient analysis (Campbell et al., 2015; Cassidy and Jordan 2011; Skeffington et al., 2015; Crockford et al., 2017), and this is something that should be considered in relation to monitoring high status river sites. This is especially relevant given the afore mentioned ASPT observations, and that low increases in P concentrations, that may go otherwise undetected, have the potential to be much more damaging in HSWs in comparison to already eutrophic systems (White et al., 2014).

Seasonal differences for PSI, E-PSI and ASPT were found in this study, with Spring scores being greater than Summer scores. This contrasts with Glendell et al. (2014), who did not find any seasonal differences between PSI scores, although a difference between years was observed. Poole et al. (2013) found PSI scores were higher in Autumn in comparison to Spring scores, while ASPT scores were higher in Spring. The decreasing PSI (and E-PSI) Summer scores in this study were primarily driven by a reduction in sensitive taxa (Group A and B – see Figures 4.4 and 4.5), with Tables 4.5 and 4.6 also conveying shifts from minimally sedimented/unsedimented to slightly sedimented during this period. This may partially be explained by life cycle strategies, with for example, the group A taxa *Rhithrogena* sp., which are univoltine and over-winter as larvae before emerging as adults in the Summer months (Elliot and Humpesch, 2012), occurring in high numbers during Spring samples in this study, but with seldom occurrence in Summer samples. Similarly, *Isoperla grammatica*, which were again prominent in Spring samples but relatively absent in Summer, emerge as adults during Summer months in Ireland, although nymphs may occur for two Summers and the overwintering prior to emergence (Feely et al., 2016). However this requires further investigation to fully appreciate the observed seasonal differences.

Seasonal differences for three and four of the physical sediment variables were also observed in 2016 and 2017, respectively. With the exception of Tile, Spring sediment levels were greater than Summer levels. Sherriff et al. (2018) highlights some evidence of seasonal increases in sediment, which were attributed in part to extreme rainfall events (but see Thompson et al., 2014). Seasonal variation in sedimentation is important, especially in relation to life-cycle stages of aquatic biota. For example, excessive suspended sediment during redd construction and egg incubation periods for spawning salmon, is likely to be more detrimental, than for the same increase in sediment occurring during winter (Bilotta and Brazier, 2008). The lower biotic metric scores in Summer along with the lower Summer sediment levels, again adds weight to the influence of life cycle strategies as the reason for seasonal discrepancies in biotic metric scores observed in this study. Finally, the results of this study may serve as a baseline with which to compare future sediment/invertebrate analysis especially in relation to HSWs in Ireland.

4.5. Conclusions

This study found that, although HSWs are pre-dominantly made up of taxa which are sensitive to sediment, for two sediment specific metrics, the PSI and E-PSI, significant differences were observed between sites that Lost status and those that Maintained status, implying that sedimentation is impacting on macro-invertebrates at some sites. The lack of any difference between Lost and Gained sites, and the lack of a difference in the historical data-set for sites that had Lost status, however, leaves an important caveat. With the exception of one sampling period, no relationship was observed for a third metric the CoFSI, which may need to be re-assessed for use in Irish HSWs. Contrastingly, although weak to moderate relationships were observed between PSI, E-PSI and the physical sediment variables, no difference between status categories for any of the physical sediment variables was observed, although this may be related to the sampling resolution. Additionally, Chapter 4 highlighted the potential for multiple-stressors, such as the interaction between sediment, organic pollution and streamflow alterations as assessed by the LIFE metric, to contribute to deteriorations in status. In contrast to the ASPT scores however, the nutrient sampling indicated little or no evidence of nutrient enrichment at the majority of sample sites, although random one off nutrient sampling as conducted in this study is likely to yield errors. Nutrient analysis at HSWs may therefore be better served by high resolution water quality monitoring. Finally, although seasonal differences were observed in this study, a likely explanation for this is the life cycle characteristics, specifically adult emergent times, of certain taxa, although this may require further investigations.

Chapter 5

5. Synthesis, Conclusions and Recommendations

5.1. Overview

High status water-bodies (HSWs) are minimally impacted water-bodies that contribute significantly to catchment diversity, as well as providing ecosystem services and recreational facilities (Ní Chatháin et al., 2012). While the protection of HSWs should be prioritised (Doody et al, 2014), as highlighted in the "High status water-bodies in the European Union chapter" (Chapter 1), to date they have received very little attention. Ireland had a high number of HSWs relative to other EU countries (data extracted from the WISE WFD database - EEA, 2015). However, in recent years a general long-term negative trend in the number of HSWs in Ireland has been observed, albeit with improvements and dis-improvements within this trend (RBMP, 2018). For example, 32 % of all Irish rivers were classified as high status in 1987-1990, compared to 18 % in 2013-2015 (RBMP, 2018). While factors such as point source pollution, septic tank emissions, sheep-dip run-off, drainage, fertilizer addition and increasing sediment pressures from agricultural, forestry and industrial sources, have been attributed to these declines, few studies have assessed these relationships (but see Conroy et al., 2016; Roberts et al., 2016; and Jiménez et al., 2018). This study aimed to address some of the knowledge deficits relating to HSWs, by providing an assessment of the current standing of HSWs in Europe and focusing on three key factors that have been attributed their declines: land use and land cover change, sedimentation and flow alteration.

The key land use and land cover change findings indicate that anthropogenically influenced land use and land cover types were linked to declines in status, with a higher level of natural/semi-natural land occurring in Maintained catchments. For example, in the period 2006-2012, land changing from Forestry to Heterogeneous Agricultural areas, increased the likelihood of Lost status occurring by 17.5 times. In contrast, land remaining as Forestry or Inland Wetlands, and therefore reducing the potential impacts of, for example, associated drainage or fertilizer addition, reduced the chance of Lost status occurring by 15 % and 4 %, respectively. This is consistent with the findings of Roberts et al. (2016), who found agricultural land use (primarily grassland) adjacent to high status water-bodies, reduced the likelihood of sites maintaining high status. However, the similarity of land use and land cover trends between sites that have Lost and Gained status provided further research questions.

To date no study has assessed the relationship between declines in HSWs and changes in hydrometric / streamflow patterns, and Chapter 3 (Streamflow Chapter) set out to address this deficit. While the results found here indicate that streamflow is not likely to be a strong factor leading to deteriorations, it has been cited as a potential reason for declines in the second cycle of the River Basin Management Plans (RBMPs) (RBMP, 2018). As with the land use and land cover change chapter, the similarity of streamflow trends between sites that have Lost and Gained status again provides an important caveat. This caveat between Lost and Gained status categories was again evident in Chapter 4 (Sediment Chapter). Using sediment specific metrics, the PSI and E-PSI, significant differences were observed in the Chapter 4 study between sites that Lost status and those that Maintained status, suggesting sediment pressures are a factor contributing to declines in high status river sites. Contrastingly however, no difference between status categories for any of the physical sediment variables was observed.

As highlighted in the literature review chapter, management strategies for preventing deteriorations (although not exclusively for HSWs), have been proposed in the Programme of Measures (POMs) of the first cycle of RBMPs. These proposals included measures targeting point source pollution from urban waste-water treatment plants, and employing treatment systems and septic tanks to manage un-sewered discharges (WRBMP, 2010; NWIRBMP, 2010; SWRBMP, 2010). The first cycle of RBMPs also proposed the use of Good Agricultural Practice Regulations (SI 101 of 2009) and Nitrate Regulations to target agricultural pollution, while other measures included targeting pollution from forestry, pesticide use, pressures from aquaculture and peat extraction, and the sale of invasive non-native species (INNS). Flood control and assessing potential future impacts of climate change are also mentioned, while local authorises and the use of Pollution Reduction Programmes (PRP) were charged with implementing measures targeting Natura 2000 sites, with particular regard to the Freshwater Pearl Mussel (*Margaritifera margaritifera*) and shellfish waters (WRBMP, 2010; NWIRBMP, 2010; SWRBMP, 2010). Additionally, for each water body a Water Management Unit Action Plan was drawn up that further stipulates necessary measures to be implemented.

Additionally, the EPA funded "Management Strategies for the Protection of High Status Water Bodies" report by Ní Chatháin et al. (2012), suggested key management strategies to protect HSWS. This report attributed declines in HSWBs to point source pollution and unintentional discharges, along with low intensity pressures such as land–use change and associated drainage and/or use of fertilizers, malfunctioning septic tanks, forestry practices, construction and development works (especially associated with wind farms), livestock accessing rivers and streams, and sheep dip pollutants. Management strategies suggested by Ní Chatháin et al. (2012) (again as highlighted in the literature review chapter) include: the use of GIS to define high status catchment borders, and incorporating this into planning and decision making processes; establishing a spatial network of high status sites following the example of the Habitats Directive (OJEC, 1992) for protected habitats, and where possible the restoration of high status sites that have deteriorated, especially in areas with large numbers of declines; and adopting additional measures from other EU Directives outside of the Water Framework Directive (WFD) (OJEC, 2000). Identifying potential pressures via the use of catchment walk-overs, screening planning applications and recording drainage networks occurring within high status catchments have also been suggested, along with knowledge-transfer programmes, and increased frequency of sampling.

Given the propensity for low intensity pressures to impact on HSWs (Ní Chatháin et al., 2012; White et al., 2014), regulations that are aimed at high-intensity activities from agriculture, forestry or peat extraction, are likely to be insufficient to mitigate against deteriorations (Roberts et al., 2016). However, the second cycle of the RBMPs (RBMPs, 2018) builds on the previous proposals, placing a much greater emphasis on preventing deteriorations and “protecting and restoring” high status water-bodies than that of the first cycle of RBMPs. The RBMP (2018) reports that there are currently 127 water-bodies (112 rivers) at risk of not achieving their objective of high ecological status, with significant pressures coming from forestry (51 sites or 40%), hydro-morphology (43 sites or 34%), agriculture (35 sites or 28%), peat extraction (16 sites or 13%) and domestic waste-water (13 sites or 10 %). The main pressures from: forestry are associated with sedimentation following clear-felling, drainage, and afforestation and establishment practices; hydro-morphological pressures are related to habitat modification and alteration of flow regimes due to land drainage, physical infrastructures e.g. (dams, and weirs) and overgrazing; agricultural pressures are related to nutrient enrichment and sedimentation from diffuse run-off, along with point source pollution from farmyards; peat extraction pressures occur due to the release of suspended solids and ammonia and hydrological modifications especially associated with drainage practices; and domestic-waste water pressures are associated with single-dwelling septic tank systems and unlicensed urban waste-water treatment plants (RBMP, 2018).

For each of these threats the RBMP (2018) highlights actions that are being taken in order reduce pressures on HSWs. For example, measures aimed at tackling agricultural pressures include the establishment of an “Agricultural Sustainability Support and Advisory Programme” that aims to employ 30 new “Advisors” to assist farmers in bringing about behavioural changes. The use of Local Authorities to mitigate stressors at a local level; the use of agri-environmental schemes such as the Green, Low-Carbon, Agri-Environment Scheme (GLAS), which contains measures specifically related to improving water quality and prioritises applicants from high status catchments; a continuation of measures relating to the use of Good Agricultural Practice Regulations and Nitrate Regulations as suggested in the first cycle of RBMPs; and knowledge transfer programmes; are also suggested.

Forestry policies have also been updated to include measures aimed at water-protection, with the Department of Agriculture, Food and the Marine (DAFM) presenting documents on "Forests and Water - Achieving Objectives under Ireland's River Basin Management Plan 2018-2021" (DAFM, 2018a), "Forestry and Freshwater Pearl Mussel Requirements" (DAFF, 2008), and "Forestry and Aerial Fertilisation Requirements" (DAFM, 2015), along with older documents such as the "Forestry and Water Quality Guidelines" (DMNR, 2000). The Forestry and Water document is entwined with the RBMP and includes measures that among others: integrate funding for forestry with that of water protection; defines the land-types that are suitable for forestry; and stipulate the environmental requirements for afforestation and deforestation (DAFM, 2018a). The Forestry and Freshwater Pearl Mussel Requirements specify the use of buffer zones, sediment traps and brash management to manage threats from, for example, sedimentation and nutrient enrichment, from forests within six km of water-bodies containing freshwater pearl mussel (DAFF, 2008); while the Aerial Fertilisation Requirements specify a minimum exclusion zone of 50 metres between an aquatic zone and applications of aerial fertilizer (DAFM, 2015).

More specifically for HSWs, the RBMP (2018) lists five principle actions that include: 1) continuing to promote and prioritise agri-environment schemes, forestry schemes and inspections of domestic waste-water treatment systems (DWWTSs) in HSWs catchments; 2) providing grant assistance to improve DWWTSs that are potentially impacting on HSWs occurring in Areas of Action (prioritised areas to be targeted with promoting best agricultural practice); 3) the use of a "Blue Dot Catchment Programme" to establish a network of HSWs with a shared agenda for the protection and restoration of HSWs; 4) establishing a working group of all relevant stakeholders for the “Blue Dot Catchment Programme”; and 5) an application for funding through the EU LIFE Integrated Project with the intention of protecting and restoring HSWs.

5.1.1. Land cover change

Approximately 64% of land in Ireland is under agriculture with a further 10.6 % under forestry (DAFM, 2018b). This compares to the European average of 40 % and 42.4% for agriculture and forestry, respectively (Eurostat, 2018), although Irelands’ national policy is to increase the forest cover to 18 % by 2046 (DAFM, 2018b). Chapter 2 (Land cover chapter) found that a higher proportion of land at Maintained status sites remained as Inland Wetlands, Scrub, and Forest over the three land cover change periods (2006-2012, 2000-2006 and 2000-2012) in contrast to Lost sites which had a higher proportion of land remaining as Pastures and Arable Land. Roberts et al. (2016) similarly found agricultural land (grasslands) to have a negative influence on sites maintaining high status, while forestry was found to have no effect. Although the RBMP (2018) highlights forestry and peatland as threats to 51 and 16 “current” high status sites, respectively, these threats are associated with afforestation and deforestation in the case of forestry, and harvesting of peat in the case of peatland. It is likely that the Forestry and Inland wetlands (primarily raised bog and blanket bog) land associated with Maintained sites in the Land Cover Change chapter, have limited anthropogenic associations, although this requires further analysis. However, should clear-felling of forestry and drainage of inland wetlands occur in these catchments classified as Maintained, this may result in future deteriorations in status. Mitigation measures, such as those specified in the RBMPs and the DAFM documents on forestry and water, to counter-act the potential threats from sedimentation and drainage, should therefore be implemented at current high status sites, prior to the commencement of any potentially detrimental actives.

The similarity of land cover findings between Lost and Gained status sites in the Land Cover Change chapter requires further investigation, especially as measures being implemented in catchments with Gained status may be replicated and possibly used to improve conditions at Lost status sites. Future studies should therefore assess if the attained Gained status is related to taxa resilience, frequency of sample monitoring (Q-value), or actual land use management practices. In any case it may be wise to implement mitigation measures at all high status, including those that have deteriorated, with the intention of reversing observed declines. Where policy measures are to be implemented, Micha et al. (2018) suggests best practice is to: engage with farmers using a bottom-up approach, that is receptive to local social and cultural behaviours; actively seek farmer participation, rather than expect voluntary engagement; provide better knowledge transfer between researchers and advisors so that advisors are better equipped to transfer knowledge onto farmers; and promote greater engagement between researchers and farmers which may facilitate a greater exchange of ideas.

5.1.2. Streamflow pressures

While Chapter 3 (Streamflow Pressures chapter) found that for most sites changes in streamflow patterns were not likely to be a major factor leading to deteriorations, there is the potential at some sites, especially in combination with other stressors, for streamflow alterations to be problematic. Additionally, while abstractions and/or droughts were not considered to be a problem at any of the assessed sites, this may change if future climate change predictions are realised. The RBMP (2018) for example, as previously mentioned, currently lists 43 high status river sites (or 34% of the 127 sites at risk of not achieving high status) as being susceptible to hydro-morphological pressures.

During the first river basin management planning cycle, along with measures proposed in the RBMP, three other legal frameworks: Regulation of Domestic Waste-Water Treatment Systems (S.I. 2033 0f 2012); Pesticide Regulations (S.I. 155 of 2012 and S.I. 159 of 2012); and Environmental Impact Assessment (Agriculture) (EIA) (S.I. 456 of 2011); were entered into law (RBMP, 2018). The EIA regulations require farmers to carry out an EIA if: 1) rural land holdings are being restructured; 2) intensive agriculture is about to commence on previously uncultivated or semi-natural land; or 3) land drainage works are about to commence. Additionally for an EIA to be carried out, for all three activities, the proposed works must: a) exceed guideline values (15 ha); or b) occur in or are likely to impact on a proposed Natural Heritage Area or Nature Reserve; or c) significantly impact on the environment (RBMP, 2018; Paul et al., 2018). With regard to carrying out EIA assessments, it may be wise to include "impacting on high status water-bodies" (especially relating to drainage) as an additional category requiring EIAs.

Where drainage is known to be problematic, leading to altered flow rates, perhaps infilling, or drain blocking should be considered. However, contrastingly the lack of drainage at some sites, especially those with heavy soils may result in excessive run-off problems (Roberts et al., 2016), with the general consensus being that drainage facilitates the absorption and fixation of P to soils (Gramlich et al., 2018). Again, carrying out EIAs in high status catchments prior to the commencement of drainage works, that takes into account local topography, soil characteristics (e.g. organic or mineral soils, percentage of sand, silt, clay, etc.), the cumulative effects of drainage within a catchment, local climate, the type of drainage to be employed and land management (see Gramlich et al., 2018), may facilitate best judgement.

The Lotic-invertebrate Index for Flow Evaluation (LIFE) method as described by Extence et al. (1999) was employed in the Streamflow Pressures chapter study to assess if streamflow alteration has impacted on river biology at high status river sites. While the LIFE index provides a general overview of streamflow as a pressure on invertebrates, its utilization to date has primarily focused on the impacts of reduced flows resulting from abstraction and drought pressures (e.g. Extence et al., 1999; Bradley et al., 2017; Westwood et al., 2017). In the Flow chapter study, while differences were found in LIFE scores between sites classified as Lost and Maintained, all scores were generally at the higher end of the scoring range and suggestive of invertebrate taxa with a preference for medium/high streamflow rates. Following on from this, perhaps a new index that is more focused on assessing the impacts of drainage should be developed, although this may prove difficult given that drainage increases winter streamflow due a more rapid escape of precipitation, and reduces summer streamflow due to the reduction of stored water (Blann et al., 2009).

Although the Streamflow chapter did not find abstractions and/or droughts (or similar patterns) to be a problem at any of the assessed sites, the impacts of climate change and potential future decreases in summer and autumn streamflows in Ireland (Murphy and Charlton, 2006; Steele-Dunne et al., 2008; Hall and Murphy, 2010; Roudier et al.,

2016) may be a cause for concern. With regard to this, it may be worth utilising the recently developed 'Drought Effect of Habitat Loss on Invertebrates' (DEHLI) index, an index that is specifically designed to assess the impacts of droughts on river systems (Chadd et al., 2017). At the very least, studies utilising this index may provide baseline data with which to compare and assess future potential impacts of climate change and/or drought/abstraction/reduced summer flow pressures.

5.1.3. Sediment pressures

Chapter 4 (Sediment chapter) found that invertebrate taxa which are sensitive to sediment were dominant at most of the sites analysed. However, for two sediment specific metrics, the Proportion of Sediment-sensitive Invertebrates (PSI) index and the Empirically-weighted PSI (E-PSI) index, significant differences were found between sites that had Lost and Maintained status. As scores at Maintained sites were higher, it is likely that sedimentation is more of a stressor at Lost sites. However, no difference was observed between sites that Lost status and those that Gained in status, thereby again leaving an important caveat. Given that agriculture and forestry are two of the main sources of fine sediment to streams (Collins and Anthony, 2008; Wagenhoff et al., 2011; Benoy et al., 2012; Thompson et al., 2014; dos Reis Oliveira et al., 2018), and both are cited by the RBMP (2018) as potential sources of pressure for 35 and 51 current high status river sites respectively, mitigation measures to combat impacts should focus on these sectors. In the UK for example, 70 % of sediment loadings to rivers has been estimated to come from agriculture, with this potentially rising to 90 % in Scotland (Rickson, 2014). This loss of sediment not only impacts on water quality, but also results in a loss of valuable soil to landowners.

Soil erosion from either tillage or land mismanagement; livestock - poaching, accessing the river-bed or grazing the riparian bank-side; and bank-side erosion are primary sources of sediment, with regard to agriculture (Waters, 1995; Lefrançois et al., 2007; Larsen et al., 2011; Benoy et al., 2012; Conroy et al., 2016). Switching from pasture to arable production, and growing “erosive crops” such as potatoes, spring grown and winter cereals, and maize fodder (Rickson, 2014) are also of concern. This situation is likely to be exacerbated by increased extreme events as a result of climate change (Scheurer et al., 2008).

Rickson (2014) in a review of the literature reported “constructed waterways (effectiveness = 83%); mulching/crop residue management (66%); wetland features (65%); edge-of-field buffer strips (64%); minimal cultivation systems (62%); and in-field grass buffer strips (61%)” as being the most effective for preventing soil loss/erosion control. Of the assessed measures “buffer strips (mean effectiveness = 54–65%), mulching/crop residue management, leaving autumn seed beds rough (13%) and allowing field drainage to deteriorate (6%)” require the least effort to implement, although the most cost-effective mitigation measures (i.e. provides most effective soil erosion at lowest cost) were determined to be management of tramlines, contour ploughing, mulching and the use of riparian buffer strips (Rickson, 2014).

For both agriculture and forestry, measures outlined in the RBMP (2018) and DAFM forestry water documents, which include the use of riparian buffer strips, should be implemented to target sediment pressures. Poole et al. (2013), for example, suggests that agri-environmental schemes focusing on maintaining woodland within a 500m buffer zone in the upper reaches of the river, may improve water quality, along with reducing the impacts of sedimentation. Davies et al. (2009) similarly highlights how agri-environmental schemes may enhance the protection of aquatic biota, although they may need to target strategic locations, as opposed to being applied with a broad brush. Targeting critical source areas (CSAs), which are comparatively minor areas within a field, farm or catchment that contribute the majority of pollutant transfers (e.g. nutrients) to aquatic systems (McDowell et al., 2014), through, for example, the use of Geographic Information System (GIS) modelling, may allow for this. Doody et al. (2012) suggests targeting CSAs as a cost effective method to counteract diffuse P pollution in Ireland, especially in HSW catchments. As there is a significant interaction between sediment transfer and nutrient enrichment of water-courses, many mitigation measures that are applicable to combating nutrient enrichment, are also applicable to mitigating against sedimentation (Rickson, 2014). Combining CSAs with the use of extended riparian buffer zones may therefore allow for the removal/absorption of the majority of pollutant stressors with the minimal of effort. This may help manage the impacts of multiple stressors especially as sediment transfer is being increasingly recognised as the source of pollutants, such as heavy metals (Carter et al., 2006), pesticides (Singh et al., 2007), and nutrients (Ballantine et al., 2009), entering aquatic systems (Walling and Collins, 2008). For example, due to selective chemical weathering (eroding), sediment entering water-courses, has the potential to contain more concentrated levels of nutrient pollutants such as P, in comparison to *in-situ* un-eroded soil (Rickson, 2014). Other examples include, agriculture influencing the amount and composition of dissolved organic matter entering aquatic systems (Graeber et al., 2012), which, as with sedimentation, has potential “cascading” consequences for the aquatic food-web (Tank et al., 2010).

At present EU legislation regarding suspended sediment concentrations is minimal (Thompson et al., 2014). The EU Freshwater Fish Directive (OJEC, 2006) stipulated that, with the exception of flooding events, suspended solids should not exceed 25 mg/L in salmonid and cyprinid waters, although this directive was repealed in 2013 in favour of the WFD (Thompson et al., 2014). However, as highlighted in the Sediment chapter, this single value covering a large range of habitat types is too simplistic, and fails to account for the exposure time of taxa to sediments, or variations in streamflow rates (Bilotta and Brazier, 2008). For example, several studies highlight how sediment levels vary between sites of differing habitat characteristics such as slope, altitude, stream gradient, stream order, etc., (Relyea et al., 2011; Bilotta et al., 2012). There is therefore a requirement to adjust the sediment guidelines to better reflect these variations between each habitat "type".

Both the Streamflow and Sediment chapters highlighted the need to distinguish between seasonal changes in invertebrate communities occurring as a result of life history traits, and changes occurring as a result of differing seasonal physical pressures, such as reduced streamflow in summer. For example, the absence of *Rhithrogena* sp. from Summer samples, while being present in high numbers during Spring samples, is likely more related to life cycle strategies, than actual streamflow or sediment pressures. Additionally, for both the Streamflow and Sediment chapters there was a requirement to urge caution in relation to the interpretation of results relating to the use of EPA Q-value (macro-invertebrate) derived status categories (Maintained, Lost and Gained) and the invertebrate metrics employed (LIFE, PSI, E-PSI etc.). While there may be some scope for movement, given that the EPA Q-value system is more focused on assessing general/organic pollution trends, and the LIFE and PSI indices are specifically related to assessing the sensitivity of invertebrate taxa to streamflow and sedimentation pressures, respectively, future projects should be better designed to insure complete independence between data-sets, so as to remove the potential for errors in data analysis.

Furthermore, invertebrate metrics that incorporate abundances, are potentially susceptible to the impacts of invasive species. Mathers et al. (2016), for example, found that LIFE and PSI metric scores were significantly elevated in comparison to control sites, following the invasion of sites by the non-native signal crayfish (*Pacifastacus leniusculus*), there-by giving a response that is not truly related to the impacts of hydrological or sedimentation pressures. While there has not yet been any recording of signal crayfish in Ireland, there has been five incidents of crayfish plague, which signal crayfish are known to transmit (Invasive Species Ireland, 2018). Of the invertebrate samples analysed in the Streamflow and Sediment chapters, the non-native New Zealand mud snail (*Potamopyrgus antipodarum*) was present in forty-two of the samples collected in Summer 2017, occurring across all status categories. Although this species is considered a medium impact invasive species (NBDC, 2018) and has a widespread distribution through-out Ireland (Anderson, 2016), at relatively low densities it has been shown not to alter surrounding invertebrate communities (Murria et al., 2008). However, its impact at high densities has been compared to that of the zebra mussel (*Dreissena polymorpha*) (Alonso and Castro-Díez, 2008). Given that it was found to be dominant at several of the sites analysed in the Streamflow and Sediment chapters, its impact on high status sites requires further investigation.

While the use of invertebrates for assessing the impacts of pollution has been commonplace for a number of years (e.g. BMWP index – Hawkes, 1998), it is only relatively recently that specific metrics such as the LIFE (Extence et al., 1999) and PSI (Extence et al., 2013) and E-PSI (Turley et al., 2016) metrics for assessing streamflow and sediment pressures, respectively, have been developed. To date the use of such metrics has been relatively rare in Ireland. Following on from this, the use of these metrics (LIFE, PSI, E-PSI and Co-FSI) should at the very least provide a baseline with which to compare any future collected data. This should allow for a more long-term assessment of how streamflow and sediment patters are changing over time. One limitation of the Streamflow and Sediment studies was the relative isolation of sampling data. In contrast to analysis carried out by, for example, Monk et al. (2012), Bradley et al. (2017) and Westwood et al. (2017), in which data-sets covering up to 30 consecutive years of LIFE scores were analysed, only two consecutive years of sampling data, with an additional two non-consecutive years of historical data was available for analysis here. Finally, given that many of these sites continue to be high status, they may serve as reference sites for other similar “type” sites, with which to compare the land use and land cover, streamflow and sediment characteristics.

5.2. Conclusions

This project set out to provide a perspective for HSWs in the EU and to assess if environmental changes resulting from changing land use and land cover trends, streamflow and sedimentation pressures were factors leading to declines in HSWs in Ireland. The main conclusions from these four areas of research are:

HSWs are sensitive areas that require special attention. However, across the E.U., while some countries have developed strategies for their protection, other countries appear more concerned with achieving the good status objective. In countries where measures for the protection of high status waterbodies have been proposed, lag times between implementing management strategies and seeing actual benefits, make assessing the effectiveness of such measures difficult. On the other hand, ignoring HSWs to concentrate on the good status objective may in the long run prove counter-productive, especially as any deteriorations may be impossible to reverse, or at least require a large time commitment. Countries that have developed strategies may benefit from the sharing of knowledge, and some examples of this are provided in Chapter 1. In Ireland a large number of deteriorations have been observed, and along with point source pollution or unintentional discharges, these deteriorations have potentially been attributed to low intensity practices such as: land–use change through drainage or fertilizer addition; and the impacts of hydrological modifications and sedimentation.

Understanding the relationship between changes in land use and land cover trends and water-bodies is crucial for mitigating against potential stressors, and together with an evaluation of both the biological and chemical status, provides a more holistic view of factors leading to deteriorations. Chapter 2 (Land Cover Change) demonstrated potential methods to carry out such a land use and land cover assessment, and found that land cover changes were linked to declines in water body status, through the overriding occurrence of anthropogenically influenced land (in comparison to the higher level of natural/semi-natural land occurring in Maintained sites). However, the similarity of land cover trends between Lost and Gained status provides further research questions especially relating to the transferability of potential resilience measures being implemented in catchments with Gained status, and the adequacy of the ecological sample survey method for detecting true changes in status. Based on this, the need for future studies to assess the influence of management strategies, land use intensity, scale and sampling error/frequency were highlighted.

Chapter 3 (Streamflow chapter) found that despite differences in LIFE scores between the status categories Lost and Maintained, scores at all sample sites were generally in the same range and indicative of rivers hosting invertebrate communities with a preference for medium/high streamflow rates. Similarly, the EPA historical data-sets indicated LIFE scores were again generally in the same range and associated with medium/fast streamflows. In contrast, the LIFE scores did not indicate abstractions and/or droughts to be a pressure at any of the sampled sites. However, in light of future climate change predictions, this may require future re-appraisals, for example through the use of the DEHLI index. The overall conclusion from Chapter 3 was that for most sites streamflow alterations are not likely to have been a major factor leading to deteriorations to date. However, for certain sites, and potentially in combination with other stressors, changes in streamflow patterns may be problematic.

Chapter 4 (Sediment chapter) found that, macro-invertebrate taxa occurring in HSWs were pre-dominantly sediment sensitive taxa. However, for two sediment specific metrics, the PSI and E-PSI, significant differences were observed between sites that Lost status and those that Maintained status, implying that at some sites, sedimentation is impacting on macro-invertebrates. As in the Land Cover Change chapter, the lack of any difference between Lost and Gained sites, leaves an important caveat, and questions that require further analysis. While weak to moderate relationships were observed between PSI, E-PSI and the physical sediment variables, no difference between status categories for any of the physical sediment variables was observed, although this may be related to the sampling resolution. Additionally, Chapter 4 highlighted the potential for multiple-stressors, such as the interaction between sediment, organic pollution and streamflow alterations as assessed by the LIFE metric, to contribute to deteriorations in status. In contrast to the ASPT scores however, the nutrient sampling indicated little or no evidence of nutrient enrichment at the majority of sample sites, although random one off nutrient sampling as conducted in this study is likely to yield errors. Nutrient analysis at HSWs may therefore be better served by high resolution water quality monitoring. Finally, both Chapters 3 and 4 found seasonal differences between invertebrate data-sets, and while a likely explanation for this is the life cycle characteristics, specifically adult emergent times of certain taxa, this may require further investigations.

5.3. Recommendations

Based on the findings of this project the main recommendations are:

- Mitigation measures, such as those specified in the RBMPs and the DAFM documents on forestry and water, to counter-act the potential threats from sedimentation and drainage, should be implemented at current high status sites, prior to the commencement of any potentially detrimental actives, such as deforestation.
- Given the similarity of findings between Lost and Gained status sites, further investigations should be conducted to assess if measures being implemented in catchments with Gained status may be replicated and possibly used to improve conditions at Lost status sites.
- It may be wise to include “impacting on high status water-bodies” as an additional category requiring EIAs (especially in relation to drainage works).
- Following the example of the LIFE and (DEHLI) index, a new index that is more focused on assessing the impacts of increases in streamflow should be developed, to allow for a more detailed assessment of the impacts of drainage on a riverine system.
- For both agriculture and forestry, measures outlined in the RBMP (2018) and DAFM forestry water documents, should be implemented to target sedimentation, with an emphasis being placed on the creation of riparian buffer zones in CSAs so as to target the majority of pollutant stressors with the minimal of effort.

- There is a requirement for better legalisation regarding the suspended sediment levels occurring in a water-body to better reflect variations in river habitat "types".
- The use of high resolution monitoring for nutrient analysis is something that should be considered, especially given the potential for low increases of P that may be undetectable by one-off random sampling, to have a relatively large impact on HSWs.
- There is a requirement to determine if the seasonal changes observed in invertebrate communities are occurring as a result of life history traits, or because of differing seasonal physical pressures, such as reduced flow in summer.
- The impact the non-native New Zealand mud snail (*Potamopyrgus antipodarum*) is having on high status sites requires further investigation.
- Finally, the use of invertebrate metrics employed in the Life and Sediment chapters (i.e. LIFE, PSI, E-PSI and Co-FSI) should prove useful as a baseline with which to compare any future collected data.

Appendix A

Supplementary data from Chapter 2 - The relationship between land cover change and high status water-bodies

Table A.1. Mann-Whitney U test results for Maintained against Gained for the period 2000-2006.

2000-2006 - Maintained vs. Gained	Mann-Whitney U	Wilcoxon W	Z	Asymp. Sig. (2-tailed)
Arable land - Arable land	1612	5440	-2.199	0.028
Arable land - Pastures	1569.5	5397.5	-2.755	0.006
Heterogeneous agricultural areas - Pastures	1644	5472	-1.964	0.049
Industrial, commercial and transport units - Industrial, commercial and transport units	1870.5	5698.5	-2.400	0.016
Industrial, commercial and transport units - Pastures	1870.5	5698.5	-2.400	0.016
Inland wetlands - Inland wetlands	1460.5	2541.5	-2.557	0.011
Pastures - Pastures	1438	5266	-2.668	0.008
Pastures - Urban fabric	1831.5	5659.5	-2.072	0.038
Urban fabric - Urban fabric	1782	5610	-2.514	0.012

Table A.2. Mann-Whitney U test results for Gained against Lost for the period 2000-2006.

2000-2006 - Gained vs. Lost	Mann-Whitney U	Wilcoxon W	Z	Asymp. Sig. (2-tailed)
Heterogeneous agricultural areas - Forest	709	1489	-2.431	0.015

Table A.3. Mann-Whitney U test results for Maintained against Lost for the period 2000-2006. * not significant at 0.05 but close to.

2000-2006 - Maintained vs. Lost	Mann-Whitney U	Wilcoxon W	Z	Asymp. Sig. (2-tailed)
Arable land - Heterogeneous agricultural areas	1566	5394	-2.608	0.009
* Artificial non-agricultural vegetated areas - Artificial non-agricultural vegetated areas	1584	5412	-1.955	0.051
Heterogeneous agricultural areas - Forest	1342	2122	-2.607	0.009
Heterogeneous agricultural areas - Mines, dumps and construction sites	1609.5	5437.5	-2.121	0.034
Heterogeneous agricultural areas - Urban fabric	1609.5	5437.5	-2.121	0.034
Inland wetlands - Inland wetlands	1220.5	2000.5	-2.512	0.012
Mines, dumps and construction sites - Mines, dumps and construction sites	1416	5244	-3.023	0.003
Pastures - Pastures	1111.5	4939.5	-3.093	0.002
Pastures - Scrub and/or herbaceous vegetation associations	1369	5197	-1.809	0.070
* Pastures - Urban fabric	1560.5	5388.5	-1.945	0.052
Urban fabric - Urban fabric	1475.5	5303.5	-2.759	0.006

Table A.4. Mann-Whitney U test results for Gained against Lost for the period 2006-2012.

2006-2012 - Gained vs. Lost	Mann-Whitney U	Wilcoxon W	Z	Asymp. Sig. (2-tailed)
Arable land - Forest	921.000	1551	-2.40	0.016
Arable land - Pastures	811.000	1441	-2.43	0.015
Continental waters - Pastures	992.000	3072	-2.11	0.035
Forest - Pastures	820.000	1450	-2.22	0.026
Heterogeneous agricultural areas - Heterogeneous agricultural areas	814.000	2894	-2.34	0.019
Heterogeneous agricultural areas - Inland wetlands	861.500	2941.5	-2.07	0.039
Inland wetlands - Mines, dumps and construction sites	992.000	3072	-2.75	0.006
Mines, dumps and construction sites - Heterogeneous agricultural areas	1024.000	3104	-2.37	0.018
Pastures - Forest	797.000	1427	-2.38	0.018

Table A.5. Mann-Whitney U test results for Maintained against Gained for the period 2006-2012.

2006-2012 - Maintained vs. Gained	Mann-Whitney U	Wilcoxon W	Z	Asymp. Sig. (2-tailed)
Arable land - Arable land	1991	4841	-2.832	0.005
Arable land - Forest	2036.5	4886.5	-2.699	0.007
Arable land - Inland wetlands	2166	5016	-2.207	0.027
Arable land - Pastures	1545	4395	-4.072	0.000
Arable land - Urban fabric	2250	5100	-2.189	0.029
Continental waters - Continental waters	2153	4233	-1.999	0.046
Continental waters - Inland wetlands	2150	4230	-2.169	0.030
Forest - Arable land	2212.5	5062.5	-2.456	0.014
Heterogeneous agricultural areas - Arable land	2082	4932	-2.759	0.006
Heterogeneous agricultural areas - Pastures	1942	4792	-1.979	0.048
Inland wetlands - Continental waters	2180	4260	-1.986	0.047
Inland wetlands - Inland wetlands	1598	3678	-3.391	0.001
Inland wetlands - Scrub and/or herbaceous vegetation associations	1812.5	3892.5	-2.485	0.013
Mines, dumps and construction sites - Arable land	2250	5100	-2.189	0.029
Mines, dumps and construction sites - Mines, dumps and construction sites	2133.5	4983.5	-2.406	0.016
Mines, dumps and construction sites - Pastures	2133.5	4983.5	-2.406	0.016
Pastures - Arable land	1915	4765	-3.112	0.002
Pastures - Forest	1789.5	4639.5	-2.603	0.009
Pastures - Mines, dumps and construction sites	2127.5	4977.5	-2.460	0.014
Pastures - Pastures	1450	4300	-4.026	0.000
Pastures - Urban fabric	1976.5	4826.5	-3.323	0.001
Scrub and/or herbaceous vegetation associations - Arable land	2209	5059	-2.130	0.033
Scrub and/or herbaceous vegetation associations - Inland wetlands	1850	3930	-2.327	0.020
Urban fabric - Forest	2212.5	5062.5	-2.456	0.014
Urban fabric - Heterogeneous agricultural areas	2250	5100	-2.189	0.029
Urban fabric - Pastures	1974.5	4824.5	-3.339	0.001
Urban fabric - Urban fabric	1971.5	4821.5	-3.362	0.001

Table A.6. Mann-Whitney U test results for Maintained against Lost for the period 2006-2012.

2006-2012 - Maintained vs. Lost	Mann-Whitney U	Wilcoxon W	Z	Asymp. Sig. (2-tailed)
Forest - Industrial, commercial and transport units	1237.500	4087.5	-2.07967	0.038
Heterogeneous agricultural areas - Heterogeneous agricultural areas	1015.000	3865	-1.96888	0.049
Heterogeneous agricultural areas - Industrial, commercial and transport units	1237.500	4087.5	-2.07967	0.038
Industrial, commercial and transport units - Industrial, commercial and transport units	1200.000	4050	-2.55876	0.011
Industrial, commercial and transport units - Inland wetlands	1237.500	4087.5	-2.07967	0.038
Industrial, commercial and transport units - Pastures	1200.000	4050	-2.55876	0.011
Industrial, commercial and transport units - Scrub and/or herbaceous vegetation associations	1237.500	4087.5	-2.07967	0.038
Inland wetlands - Industrial, commercial and transport units	1237.500	4087.5	-2.07967	0.038
Inland wetlands - Inland wetlands	811.000	1441	-3.21956	0.001
Mines, dumps and construction sites - Heterogeneous agricultural areas	1200.000	4050	-2.55876	0.011
Mines, dumps and construction sites - Inland wetlands	1237.500	4087.5	-2.07967	0.038
Pastures - Mines, dumps and construction sites	1160.000	4010	-2.31299	0.021
Pastures - Urban fabric	1121.500	3971.5	-2.72246	0.006
Scrub and/or herbaceous vegetation associations - Mines, dumps and construction sites	1237.500	4087.5	-2.07967	0.038
Urban fabric - Pastures	1120.500	3970.5	-2.73671	0.006
Urban fabric - Urban fabric	1120.500	3970.5	-2.73671	0.006

Table A.7. Mann-Whitney U test results for Gained against Lost for the period 2000-2012.

2000-2012 - Gained vs. Lost	Mann-Whitney U	Wilcoxon W	Z	Asymp. Sig. (2-tailed)
Heterogeneous agricultural areas - Heterogeneous agricultural areas	610	2888	-2.74091	0.006
Mines, dumps and construction sites - Heterogeneous agricultural areas	871	3149	-2.19923	0.028
Mines, dumps and construction sites - Inland wetlands	871	3149	-2.19923	0.028

Table A.8. Mann-Whitney U test results for Maintained against Gained for the period 2000-2012.

2000-2012 - Maintained vs. Gained	Mann-Whitney U	Wilcoxon W	Z	Asymp. Sig. (2-tailed)
Arable land - Arable land	1713	3604	-2.673	0.008
Arable land - Forest	1684.5	3575.5	-2.981	0.003
Arable land - Inland wetlands	1807	3698	-2.425	0.015
Arable land - Pastures	1372.5	3263.5	-3.531	0.000
Heterogeneous agricultural areas - Pastures	1555	3446	-2.383	0.017
Inland wetlands - Inland wetlands	1206	3484	-3.997	0.000
Open spaces with little or no vegetation - Inland wetlands	1876	4154	-2.381	0.017
Open spaces with little or no vegetation - Open spaces with little or no vegetation	1909.5	4187.5	-2.121	0.034
Pastures - Arable land	1656	3547	-2.867	0.004
Pastures - Forest	1477.5	3368.5	-2.726	0.006
Pastures - Heterogeneous agricultural areas	1627	3518	-2.035	0.042
Pastures - Pastures	1317	3208	-3.482	0.000
Pastures - Urban fabric	1804.5	3695.5	-2.255	0.024
Scrub and/or herbaceous vegetation associations - Pastures	1508.5	3399.5	-2.592	0.010
Urban fabric - Pastures	1738.5	3629.5	-3.127	0.002
Urban fabric - Urban fabric	1769	3660	-2.955	0.003

Table A.9. Mann-Whitney U test results for Maintained against Lost for the period 2000-2012.

2000-2012 - Maintained vs. Lost	Mann-Whitney U	Wilcoxon W	Z	Asymp. Sig. (2-tailed)
Forest - Artificial non-agricultural vegetated areas	793	2684	-2.0992	0.036
Forest - Urban fabric	793	2684	-2.0992	0.036
Heterogeneous agricultural areas - Heterogeneous agricultural areas	591	2482	-2.39771	0.016
Inland wetlands - Inland wetlands	428	834	-3.76477	0.000
Mines, dumps and construction sites - Heterogeneous agricultural areas	793	2684	-2.0992	0.036
Mines, dumps and construction sites - Inland wetlands	793	2684	-2.0992	0.036
Pastures - Pastures	629	2520	-1.99768	0.046
Pastures - Urban fabric	699	2590	-2.7602	0.006
Urban fabric - Pastures	701.5	2592.5	-3.37629	0.001
Urban fabric - Urban fabric	701.5	2592.5	-3.37629	0.001

Appendix B

Supplementary data relating to the sixty-five sampling sites used in Chapters 3 and 4.

Table B.1. List of sixty-five sample sites with name, location, Easting and Northing, stream order, authority (e.g. CE = Clare Co. Co.) and RIV-type (Hardness Slope).

Station Code	River Name	Station Location	Easting	Northing	ORDER	Authority	RIV-TYPE
25A030100	Ardcloony	Ballycorney Br	166965	170671	2	CE	22
25B070200	Bleach	Bleach Br	157008	195709	4	CE	21
25B100100	Bow	Bow River Br	166576	187099	4	CE	22
25B100200	Bow	Cloontymweenagh Br	167422	185157	4	CE	22
25B150050	Ballinlough Stream	Br S Acres	167848	202290	2	GY	32
25B150500	Ballinlough Stream	Bridge u/s Cappagh River	176786	205015	3	GY	31
25C030100	Cappagh (Galway)	Metal Bridge	168182	211277	2	GY	32
25C070200	Cloghaun	Br N of Gortacummer	152101	183895	3	CE	31
25C090100	Corra	Br SE of Corrakyle	161499	191278	3	CE	11
25C090400	Corra	Gortaderry Br	159503	189303	3	CE	12
25D100200	Derrainy	Br at Derrainy	174987	191548	3	CE	22
25G040025	Graney (Shannon)	Caher Br	155396	190000	3	CE	12
25W010300	Woodford (Galway)	Rossmore Br	177206	198435	3	GY	31
26D070700	Derrymullan Stream	1st Bridge u/s Suck confl	184114	231759	4	GY	31
26F020080	Feorish (Ballyfarnon)	Br SW Geevagh	183489	316523	3	SO	32
26F020250	Feorish (Ballyfarnon)	1.2 km d/s Ballyfarnon Bridge	186818	312960	3	RN	32
26I030300	Island	Br SW Bookalagh	166216	272970	4	GY	31
26I030400	Island	Castlerea Bridge - Ballymoe	169477	271718	4	GY	31
26K010300	Killian	Bridge u/s Shiven (S.) River	174129	249049	3	GY	31
26L030350	Lung	Bridge W. of Banada	163353	294384	4	RN	31
26S030400	Shiven (South)	Islandcausk Bridge	178709	249360	4	GY	31
26Y010200	Yellow (Ballinaglera)	Br u/s Lough Allen	199102	322001	4	LM	32
27B020600	Broadford	Near Graffa Bridge	159231	171945	3	CE	11
27G020600	Gourna	Br u/s Owenogarney R confl	148097	164129	3	CE	22

27O010700	Owenogarney	Pollagh Bridge	150042	170246	4	CE	31
29B020100	Beagh	S Cloghnakeava	146728	200612	4	GY	31
29B040300	Boleyneendorrish	Kenny's Br	151415	205622	3	GY	21
29O011000	Owendalulleegh	Br SE Killafeen	148354	197128	4	GY	21
30G010250	Glensaul	0.25 km d/s Br in Tourmakeady	109936	267975	4	MO	22
30N010100	Nanny (Tuam)	Br NW Loughpark	145518	252926	2	GY	31
31R010100	Recess	Bunskannive Bridge	93634	246316	2	GY	21
32B030050	Bunowen (Louisburgh)	Br N laghta Eighter	85176	275060	4	MO	12
32C010020	Carrownisky	Glenkeen Bridge	81858	272370	3	MO	14
32C030150	Crumpaun	N. of Lough Beltra	109004	301432	4	MO	31
32C050050	Carrowbeg (Westport)	Cloghan Bridge	101084	279825	3	MO	31
32E010030	Erriff	The Wooden Br (W of Cregganmore)	91536	274307	4	MO	12
32G070300	Glenisland	Bridge u/s Lough Beltra	107304	296716	3	MO	13
32O040250	Owennabrockagh	Br NE Derrintloura	106081	291958	3	MO	31
33A020100	Altnabrocky	Just u/s Owenmore River confl	96895	319813	4	MO	31
33B010100	Ballinglen	Ballinglen Bridge	110246	334211	4	MO	32
33G020200	Glencullin (North Mayo)	Killerduff Bridge	109339	339262	4	MO	33
33K010200	Keerglen	SW of Kilkeerglen	109269	333221	3	MO	32
33O040050	Owenmore (Mayo)	Br SE Srahnakilly	97833	323161	4	MO	31
34C030030	Cloonaghmore	Br u/s Ford SSE Tawnywaddyduff	107128	324369	3	MO	32
34C030150	Cloonaghmore	Ballintober Bridge	114375	326127	4	MO	32
34C050030	Clydagh (Castlebar)	Br NW Ardvarney	114243	296525	3	MO	12
34C100300	Cloonlavis	Bridge u/s Yellow R confluence	135750	285160	3	MO	31
34D030800	Duvowen	Br u/s Cloonaghmore River	114125	326062	3	MO	32
34G010020	Glenree	Bridge near Carrownaglogh	136084	319511	3	MO	32
34G020200	Glore (Mayo)	Glore Bridge	135000	291785	3	MO	31
34M020100	Moy	Bridge S.E. of Cloonacool	149279	316791	4	SO	31
34O030200	Owengarve (Sligo)	Dawros Br	145310	307417	4	SO	31
34S030050	Spaddagh	Br N. of Castlesheenaghan	139460	296734	2	MO	31
34T010500	Trimoge	Tullyroe Br	133005	296373	4	MO	32
34Y010100	Yellow(Foxford)	Yellow (Foxford) - Ford W. of Corlee	132280	308607		MO	12
34Y010400	Yellow (Foxford)	Bridge u/s Moy River confl	128236	306728	4	MO	12

34Y020275	Yellow (Knock)	Bridge N.E. of Faughil	137241	286818	3	MO	31
35C030200	Cashel Stream (Bonet)	Bridge W. of Corratimore	184966	327185	2	LM	31
35D050020	Duff	Br E of Cloontypruglish	179056	348081	3	LM	33
35D161000	Dunmoran	Br WNW Longford Demesne	155096	330598	3	SO	31
35F010100	Finned	Bridge E.N.E. of Rathmacurkey	136984	330873	2	SO	32
35G020200	Glenaniff	Bridge u/s Lough Melvin	192043	349681	3	LM	32
35G030100	Gowlan (Sligo)	Ford u/s Easky River confl	138828	326554	3	SO	32
35G040080	Grange (Sligo)	Lukes Bridge	169769	347329	3	SO	33
36R020200	Roo	Br W of Barran	203133	335566	2	CN	32

Table B.2. List of sixty-five sample sites with EPA sampling periods WFD status classifications from which, along with the "Final Q-vale", determined the "Status at 2016" which was used to classify the sample sites.

			EPA_sampling period			**Final Q-value used**	
Station_Code	**2001_2003**	**2004_2006**	**2007_2009**	**2010__2012**	**2013-2015**	**(year)**	**Status_at_2016**
25A030100	High	High	High	High	High	4.5 (2014)	Maintained
25B070200	High	#N/A	#N/A	High	Good	4 (2014)	Lost
25B100100	High	#N/A	Good	High	High	4.5 (2014)	Gained
25B100200	High	High	High	High	High	4.5 (2014)	Maintained
25B150050	High	#N/A	High	High	Good	4 (2014)	Lost
25B150500	High	#N/A	High	High	High	4.5 (2014)	Maintained
25C030100	Moderate	Good	High	Good	High	4.5 (2014)	Gained
25C070200	High	Good	Good	High	High	4.5 (2014)	Gained
25C090100	High			Poor	Good	4 (2014)	Lost
25C090400	High	High	High	High	High	4.5 (2014)	Maintained
25D100200	High	High	High	Moderate	Good	4 (2014)	Lost
25G040025	High	High	High	High	High	5 (2014)	Maintained
25W010300	High	High	High	Good	Good	4 (2014)	Lost
26D070700	Good	#N/A	Good	High	Moderate (3-4)	3.5 (2014)	Lost

26F020080	High	High	High	Good	High	4.5 (2014)	Gained
26F020250	High	High	High	Good	High	4.5 (2014)	Gained
26I030300	High	#N/A	High	High	High	4.5 (2014)	Maintained
26I030400	Moderate	Good	High	High	Good	4 (2014)	Lost
26K010300	Good	Good	Good	High	High	4.5 (2014)	Gained
26L030350	#N/A	Good	Good	High	High	4.5 (2014)	Gained
26S030400	Good	Good	High	High	High	4.5 (2014)	Maintained
26Y010200	High	Good	Good	High	High	4.5 (2014)	Gained
27B020600	Moderate	High	High	Good	High	4 (2016)	Lost
27G020600	Good	High	Good	High	High	4.5 (2016)	Maintained
27O010700	Good	High	#N/A	High	High	4.5 (2016)	Maintained
29B020100	Good	Good	High	Good	Moderate (3-4)	3.5 (2015)	Lost
29B040300	High	High	#N/A	High	High	4.5 (2015)	Maintained
29O011000	Good	Good	High	High	High	4.5 (2015)	Maintained
30G010250	Good	High	High	High	High	4.5 (2015)	Maintained
30N010100	Good	High	#N/A	Good	Good	4 (2015)	Lost
31R010100	Good	High	High	High	Moderate (3-4)	3.5 (2015)	Lost
32B030050	High	High	High	High	High	4.5 (2014)	Maintained
32C010020	High	High	High	High	High	4.5 (2014)	Maintained
32C030150	Good	High	High	Good	High	4.5 (2014)	Gained
32C050050	High	High	High	High	High	4.5 (2014)	Maintained
32E010030	High	Good	Good	High	High	4.5 (2014)	Gained
32G070300	High	Good	High	High	Good	4 (2014)	Lost
32O040250	High	Good	Good	High	High	4.5 (2014)	Gained
33A020100	High	Good	Good	High	Good	4 (2014)	Lost
33B010100	High	High	High	High	Good	4 (2014)	Lost
33G020200	Good	Good	Good	High	High	4.5 (2014)	Gained
33K010200	High	High	High	High	High	4.5 (2014)	Maintained
33O040050	Good	Good	High	Good	High	5 (2014)	Gained
34C030030	High	High	High	Good	High	4.5 (2016)	Gained
34C030150	Good	High	High	Good	High	4.5 (2016)	Gained
34C050030	High	High	High	High	Good	4.5 (2016)	Gained

34C100300	Good	High	High	High	Moderate (3-4)	4 (2016)	Lost
34D030800	Good	High	High	Good	Good	4 (2016)	Lost
34G010020	High	High	High	High	Good	4.5 (2016)	Gained
34G020200	Good	High	High	Good	High	3.5 (2016)	Lost
34M020100	Good	High	High	Good	Good	4.5 (2016)	Gained
34O030200	High	High	High	High	High	4.5 (2016)	Maintained
34S030050	Good	High	High	Good	Good	4.5 (2016)	Gained
34T010500	Good	Good	Good	High	High	4.5 (2016)	Gained
34Y010100	High	High	High	High	High	5 (2016)	Maintained
34Y010400	High	High	High	High	High	4.5 (2016)	Maintained
34Y020275	Good	High	#N/A	Good	High	4 (2016)	Lost
35C030200	High	High	High	High	High	4.5 (2015)	Maintained
35D050020	High	High	High	High			Maintained
35D161000	High	High	High	Moderate	Moderate (3-4)	3.5 (2015)	Lost
35F010100	High	High	High	High	Good	4 (2015)	Lost
35G020200	High	High	High	High			Maintained
35G030100	High	Good	Good	High	High	4.5 (2015)	Gained
35G040080	High	High	High	High	High	4.5 (2015)	Maintained
36R020200	#N/A	#N/A	High	High	High	5 (2014)	Maintained

Appendix C

Supplementary data relating to Chapter 3 - Investigating hydrological pressures on high status rivers.

Table C.1. List of sixty-five sample sites with information for the associated hydrometic station from which flow data was derived. The distance (Km) of the sample site from the hydrometric/flow and whether or not the hydrometric station was located on the same waterbody as the sampling site is also presented.

Station	Hydrometric station	Hydrometric station_Name	Responsibl	EASTING	NORTHING	TYPE	Distance (Km) sample site to flow station	Same waterbody
25C030100	25020	Killeen	OPW	179761	211030	River	11.6	different River on same waterbody
25B150050	25020	Killeen	OPW	179761	211030	River	14.775	No, different waterbody
25B150500	25020	Killeen	OPW	179761	211030	River	6.71	different River on same waterbody
25W010300	25020	Killeen	OPW	179761	211030	River	12.796	different River on same waterbody
25B100100	25030	Scarriff	OPW	164180	184277	River	3.657	No, different waterbody
25C070200	25030	Scarriff	OPW	164180	184277	River	12.085	different River on same waterbody
27B020600	25030	Scarriff	OPW	164180	184277	River	13.58	No, different waterbody
25C090100	25030	Scarriff	OPW	164180	184277	River	7.504	different River on same waterbody
25D100200	25030	Scarriff	OPW	164180	184277	River	13.029	different River on same waterbody
25B100200	25030	Scarriff	OPW	164180	184277	River	3.358	No, different waterbody
25C090400	25030	Scarriff	OPW	164180	184277	River	6.865	different River on same waterbody
25G040025	25030	Scarriff	OPW	164180	184277	River	10.48	different River on same waterbody
25A030100	25044	Coole	EPA	170946	169510	River	4.14	No, different waterbody
26K010300	26002	Rookwood	OPW	180656	257075	River	10.344	different River on same waterbody
26S030400	26002	Rookwood	OPW	180656	257075	River	7.9	different River on same waterbody
26I030400	26002	Rookwood	OPW	180656	257075	River	18.442	Same waterbody
26D070700	26007	Bellagill	OPW	184175	234570	River	4.5	
35C030200	26029	Dowra	EPA	199064	326947	River	14.088	No, Trib?? different waterbody - Lake??
36R020200	26029	Dowra	EPA	199064	326947	River	9.531	No, Trib?? different waterbody - Lake??

26Y010200	26029	Dowra	EPA	199064	326947	River	4.95	No - do not use - use 26SO20300
26F020080	26030	L. Allen D/S	EPA	196137	312418	River	13.141	different River on same waterbody
26F020250	26030	L. Allen D/S	EPA	196137	312418	River	9.334	different River on same waterbody
25B070200	29071	Cutra	EPA	148200	197900	Lake	9.076	No, different waterbody
29B020100	29071	Cutra	EPA	148200	197900	Lake	3.076	No, different waterbody - on Lake??
29B040300	29071	Cutra	EPA	148200	197900	Lake	8.362	No, different waterbody - on Lake??
29O011000	29071	Cutra	EPA	148200	197900	Lake	0.787	
30N010100	30007	Ballygaddy	OPW	142000	253772	River	5.8	Trib
26I030300	30020	Ballyhaunis	EPA	149616	279434	River	17.8	No, different waterbody
30G010250	30047	Keel Weir	EPA	116124	267784	River	6.19	No, Other side of lake??
31R010100	31072	Derryclare	EPA	80279	247497	Lake	13.5	No, different waterbody
32E010030	32006	Coolloughra	EPA	102279	282750	River	13.66	No, different waterbody
32B030050	32006	Coolloughra	EPA	102279	282750	River	18.75	No, different waterbody
32C050050	32006	Coolloughra	EPA	102279	282750	River	3.15	Unsure
32O040250	32012	Newport Weir	EPA	99773	294400	River	6.764	No, different waterbody
32G070300	32012	Newport Weir	EPA	99773	294400	River	7.879	No, different waterbody - + Lake Beltra
34C050030	32012	Newport Weir	EPA	99773	294400	River	14.636	No, different waterbody
32C010020	32026	Bundorragha	EPA	84136	263374	River	9.279	No, different waterbody
34Y010100	34001	Rahans	OPW	124367	317782	River	12.11	Yes, but trib diff river
34Y010400	34001	Rahans	OPW	124367	317782	River	11.7	Yes, but trib diff river
33G020200	34007	Ballycarroon	OPW	112074	315968	River	23.454	No, different waterbody
33O040050	34007	Ballycarroon	OPW	112074	315968	River	15.954	No, different waterbody
34C030030	34007	Ballycarroon	OPW	112074	315968	River	9.796	No, different waterbody
34C030150	34007	Ballycarroon	OPW	112074	315968	River	10.41	No, different waterbody
33A020100	34007	Ballycarroon	OPW	112074	315968	River	15.658	No, different waterbody
33B010100	34007	Ballycarroon	OPW	112074	315968	River	18.341	No, different waterbody
34D030800	34007	Ballycarroon	OPW	112074	315968	River	10.3	No, Trib?? different waterbody
33K010200	34007	Ballycarroon	OPW	112074	315968	River	17.495	No, different waterbody
34Y020275	34024	Kiltimagh	EPA	133333	289236	River	4.595	Trib
34C100300	34024	Kiltimagh	EPA	133333	289236	River	4.738	No, different waterbody
34G020200	34024	Kiltimagh	EPA	133333	289236	River	5	Trib
34T010500	34024	Kiltimagh	EPA	133333	289236	River	7.181	Trib

26L030350	34031	Charlestown	EPA	147725	301920	River	17.35	No, different waterbody
34S030050	34031	Charlestown	EPA	147725	301920	River	9.79	different River on same waterbody
34O030200	34031	Charlestown	EPA	147725	301920	River	6.044	different River on same waterbody
35G030100	35072	Trasgarve	EPA	144806	323780	Lake	6.590	No, Trib?? different waterbody - Lake??
34G010020	35072	Trasgarve	EPA	144806	323780	Lake	9.710	No, Trib?? different waterbody - Lake??
35D161000	35072	Trasgarve	EPA	144806	323780	Lake	12.34	No, Trib?? different waterbody - Lake??
35F010100	35072	Trasgarve	EPA	144806	323780	Lake	10.559	No, Trib?? different waterbody - Lake??
34M020100	35072	Trasgarve	EPA	144806	323780	Lake	8.297	No, Trib?? different waterbody - Lake??
27G020600	No near-by							
32C030150	No near-by							
35D050020	No near-by							
35G040080	No near-by							
35G020200	No near-by							
27O010700	No near-by							

Table C.2. List of sixty-five sample sites with information for the associated Met-Éireann rainfall station from which rainfall data was derived. The distance (Km) of the rainfall station to the hydrometric/flow station and the distance (Km) of the rainfall station to the sample site are also presented.

Station Code	Hydrometric station	Met_Eireann rainfall station	Rainfall stat. name	height (m)	Easting	Northing	Distance (Km) of rainfall stat. to Hydromet. stat.	Distance (Km) of raifall stat. to sample site
32G070300	32012	833	Newport (Furnace)	14	96700	298100	4.8	10.69
32O040250	32012	833	Newport (Furnace)	14	96700	298100	4.8	11.185
34C050030	32012	833	Newport (Furnace)	14	96700	298100	4.8	17.593
26I030300	30020	1128	Loughglinn	98	163400	286000	15.26	13
32B030050	32006	1433	Westport (Carrabawn)	56	99400	283600	3.032	16.59
32C050050	32006	1433	Westport (Carrabawn)	56	99400	283600	3.032	4.133
32E010030	32006	1433	Westport (Carrabawn)	56	99400	283600	3.032	12.148
25A030100	25044	1619	Birdhill (Parteen Weir)	34	168100	167900	3.2	2.994

26F020080	26030	1729	Drumshanbo	54	196000	312500	0.159	13.313
26F020250	26030	1729	Drumshanbo	54	196000	312500	0.159	9.184
25B150050	25020	1819	Portumna O.P.W.	35	187200	204600	9.78	19.49
25B150500	25020	1819	Portumna O.P.W.	35	187200	204600	9.78	10.406
25C030100	25020	1819	Portumna O.P.W.	35	187200	204600	9.78	20.19
25W010300	25020	1819	Portumna O.P.W.	35	187200	204600	9.78	11.78
25B070200	29071	2018	Carheeny Beg	49	144400	194300	5.341	12.684
29B020100	29071	2018	Carheeny Beg	49	144400	194300	5.341	3.1
29B040300	29071	2018	Carheeny Beg	49	144400	194300	5.341	13.312
29O011000	29071	2018	Carheeny Beg	49	144400	194300	5.341	4.8
26Y010200	26029	2037	Cuilcagh Mtns.	290	213000	324100	14.26	14.08
35C030200	26029	2037	Cuilcagh Mtns.	290	213000	324100	14.26	28.176
36R020200	26029	2037	Cuilcagh Mtns.	290	213000	324100	14.26	15.15
32C010020	32026	2426	Delphi Lodge Ii	30	84400	266000	2.567	6.858
26D070700	26007	2628	Ballinasloe (Derrymullen)	43	183400	232200	2.493	0.896
26I030400	26002	2928	Athleague	61	181800	257500	1.22	18.8
26K010300	26002	2928	Athleague	61	181800	257500	1.22	11.4
26S030400	26002	2928	Athleague	61	181800	257500	1.22	8.7
30N010100	30007	3027	Milltown	50	141000	262800	8.7	10.8
34G010020	35072	3135	Cloonacool (L. Easkey)	204	144600	320700	3.137	8.5
34M020100	35072	3135	Cloonacool (L. Easkey)	204	144600	320700	3.137	6.097
35D161000	35072	3135	Cloonacool (L. Easkey)	204	144600	320700	3.137	14.485
35F010100	35072	3135	Cloonacool (L. Easkey)	204	144600	320700	3.137	12.762
35G030100	35072	3135	Cloonacool (L. Easkey)	204	144600	320700	3.137	8.228
34C100300	34024	3335	Straide	21	126100	297900	11.28	15.982
34G020200	34024	3335	Straide	21	126100	297900	11.28	1.79
34T010500	34024	3335	Straide	21	126100	297900	11.28	7.071
34Y020275	34024	3335	Straide	21	126100	297900	11.28	15.696
34Y010100	34001	3735	Ballina (Shanaghy)	24	125600	318300	1.3	11.272
34Y010400	34001	3735	Ballina (Shanaghy)	24	125600	318300	1.3	11.4
31R010100	31072	4827	Maam Valley	58	93500	255200	15.3	8.8

30G010250	30047	4927	Tourmakeady (Water Treatment Wor	140	109200	271200	7.692	3.2
26L030350	34031	4935	Knock_Airport	201	146783	296363	5.6	16.6
34S030050	34031	4935	Knock_Airport	201	146783	296363	5.677	7.381
33A020100	34007	5035	Crossmolina (Castlehill)	17	114200	314000	2.897	18.25
33B010100	34007	5035	Crossmolina (Castlehill)	17	114200	314000	2.897	20.646
33G020200	34007	5035	Crossmolina (Castlehill)	17	114200	314000	2.897	25.832
33K010200	34007	5035	Crossmolina (Castlehill)	17	114200	314000	2.897	19.881
33O040050	34007	5035	Crossmolina (Castlehill)	17	114200	314000	2.897	18.844
34C030030	34007	5035	Crossmolina (Castlehill)	17	114200	314000	2.897	12.483
34C030150	34007	5035	Crossmolina (Castlehill)	17	114200	314000	2.897	12.4128
34D030800	34007	5035	Crossmolina (Castlehill)	17	114200	314000	2.897	12.127
34O030200	34031	5435	Curry	63	149400	306400	4.23	4.826
25B100100	25030	6819	Scarriff (Fossabeg)	61	164100	184900	0.628	3.7
25B100200	25030	6819	Scarriff (Fossabeg)	61	164100	184900	0.628	3.33
25C070200	25030	6819	Scarriff (Fossabeg)	61	164100	184900	0.628	11.95
25C090100	25030	6819	Scarriff (Fossabeg)	61	164100	184900	0.628	6.89
25C090400	25030	6819	Scarriff (Fossabeg)	61	164100	184900	0.628	6.4
25D100200	25030	6819	Scarriff (Fossabeg)	61	164100	184900	0.628	12.855
25G040025	25030	6819	Scarriff (Fossabeg)	61	164100	184900	0.628	10.43
27B020600	25030	6819	Scarriff (Fossabeg)	61	164100	184900	0.628	13.696
27G020600	n/a							
27O010700	n/a							
32C030150	n/a							
35D050020	n/a							
35G020200	n/a							
35G040080	n/a							

Table C.3. List of LIFE scores [species/mixed taxon level] occurring at the sixty-five sample sites during Spring 2016, Summer 2016, Spring 2017 and Summer 2017, with the number of scoring taxa in parenthesis.

		LIFE- Species/N (Num. scoring taxa)			
Station	**Status**	**Spring 16**	**Summer 16**	**Spring 17**	**Summer 17**
25A030100	Maintained	8.65 (34)	8.42 (19)	8.91 (23)	8.67 (21)
25B070200	Lost	8.27 (22)	8.39 (23)	8.22 (27)	8.52 (23)
25B100100	Gained	8.22 (36)	8.42 (24)	8.45 (38)	8.32 (19)
25B100200	Maintained	8.6 (35)	8.39 (23)	8.59 (34)	8 (18)
25B150050	Lost	8.13 (31)	8.08 (26)	8.45 (33)	8.05 (21)
25B150500	Maintained	8.12 (34)		9 (25)	7.67 (21)
25C030100	Gained	8.43 (28)	7.93 (15)	8.51 (35)	8.18 (17)
25C070200	Gained	8.46 (35)	8.43 (21)	8.28 (29)	8.44 (25)
25C090100	Lost	8.67 (27)	8.42 (24)	8.32 (34)	8.22 (27)
25C090400	Maintained	8.39 (33)	8.17 (24)	8.21 (39)	8.3 (27)
25D100200	Lost	8.26 (23)	8.14 (21)	8.68 (25)	8.11 (18)
25G040025	Maintained	8.52 (31)	8.48 (29)	8.52 (29)	8.36 (25)
25W010300	Lost	7.95 (38)		8.31 (36)	7.73 (22)
26D070700	Lost	8.23 (26)	8.25 (24)	8.05 (20)	8.1 (21)
26F020080	Gained	8.13 (23)	7.95 (21)	8.14 (21)	8.14 (29)
26F020250	Gained	8.13 (32)	7.92 (26)	8.45 (22)	8 (24)
26I030300	Maintained	8.18 (28)	8.07 (27)	8.29 (21)	8.06 (17)
26I030400	Lost	8.28 (25)	8.44 (27)	8.81 (26)	7.91 (23)
26K010300	Gained	8.36 (22)	8.14 (44)	8.33 (33)	8.07 (42)
26L030350	Gained	8.17 (30)	8.19 (37)	8.24 (29)	8.12 (33)
26S030400	Maintained	7.46 (24)	7.31 (29)	8.54 (28)	
26Y010200	Gained	8.3 (23)	7.92 (12)	8.39 (18)	8.14 (22)
27B020600	Lost	8.57 (28)	8 (14)	8.75 (20)	7.86 (14)
27G020600	Maintained	8.74 (31)	8.24 (25)	8.74 (31)	8.47 (19)
27O010700	Maintained	8.28 (25)	8.04 (27)	8.31 (26)	
29B020100	Lost	7.91 (32)	7.73 (33)	7.83 (29)	7.63 (30)
29B040300	Maintained	8.52 (29)	8.34 (29)	8.77 (22)	8.61 (18)
29O011000	Maintained	8.52 (29)	8.47 (30)	8.45 (29)	8.62 (26)
30G010250	Maintained	8.13 (15)	7.95 (19)	8.35 (26)	8.39 (23)
30N010100	Lost	7.9 (31)	8 (24)	8.17 (30)	7.92 (26)
31R010100	Lost	8.54 (13)		8.25 (8)	8.25 (12)
32B030050	Maintained	8.81 (21)	8.27 (11)	8.36 (22)	8 (11)
32C010020	Maintained	8.63 (19)	8.43 (23)	8.58 (26)	8.24 (17)
32C030150	Gained	8.29 (34)	8.35 (26)	8.68 (25)	7.96 (27)
32C050050	Maintained	8.5 (30)	8.17 (23)	8.67 (24)	8.48 (25)
32E010030	Gained	8.5 (28)	7.95 (22)	8.52 (33)	8.33 (21)
32G070300	Lost	8.55 (20)	8.42 (24)	8.33 (21)	8.24 (17)
32O040250	Gained	8.09 (22)	7.88 (26)	8.14 (22)	7.7 (10)
33A020100	Lost	8.09 (23)	8.2 (25)	8.18 (11)	8.04 (25)
33B010100	Lost	8.41 (29)	8.33 (27)	8.56 (34)	8 (21)
33G020200	Gained	8.67 (21)	8.33 (27)	8.25 (24)	8.06 (18)
33K010200	Maintained	8.42 (31)	8.5 (24)	8.48 (29)	8.23 (22)
33O040050	Gained	8.39 (28)	8.11 (27)	8.64 (28)	8.1 (20)
34C030030	Gained	8.43 (35)	8.37 (38)	8.57 (37)	
34C030150	Gained	8.27 (33)	8.7 (30)	8.7 (33)	8.14 (35)
34C100300	Lost	7.35 (17)	7.74 (19)	8.17 (24)	7.44 (18)
34D030800	Lost	8.64 (25)	8.31 (39)	8.46 (35)	8.21 (34)
34G010020	Gained	8.33 (24)	7.95 (19)	8.22 (27)	8 (15)
34G020200	Lost	8.28 (25)	8.47 (15)	8.3 (23)	8.13 (16)

34M020100	Gained	8.55 (29)	8.47 (30)	8.76 (25)	8.57 (21)
34O030200	Maintained	8.34 (35)	7.97 (38)	8.26 (34)	8.18 (34)
34S030050	Gained	8.11 (28)	8.2 (30)	8.48 (31)	8.48 (23)
34T010500	Gained	8.53 (32)	8.3 (27)	8.61 (33)	8.54 (24)
34Y010100	Maintained	8.39 (31)	8.28 (32)	8.56 (36)	7.96 (27)
34Y010400	Maintained	8.52 (33)	8.38 (24)	8.71 (35)	8.24 (25)
34Y020275	Lost	7.67 (24)	8.12 (17)	8.11 (27)	7.96 (24)
35C030200	Maintained	8.5 (22)	8.17 (24)	8.48 (23)	8.27 (26)
35D050020	Maintained	8.74 (23)	8.39 (23)	8.45 (31)	8.21 (19)
35D161000	Lost	8.13 (30)	8.61 (18)	8.52 (21)	8.05 (21)
35F010100	Lost	8.6 (30)	8.63 (27)	8.32 (31)	8.06 (16)
35G020200	Maintained	8.24 (29)	8.46 (24)	8.42 (31)	8.32 (25)
35G030100	Gained	8.54 (28)	8.61 (31)	8.57 (28)	8.21 (19)
35G040080	Maintained	9.04 (25)	8.72 (18)	8.69 (29)	8.6 (20)
36R020200	Maintained	8.52 (23)	8.08 (26)	8.24 (29)	8.24 (21)

Table C.4. List of LIFE scores [family level] occurring at the sixty-five sample sites during Spring 2016, Summer 2016, Spring 2017 and Summer 2017, with the number of scoring taxa in parenthesis. *Please note these family level scores were not used in any analysis and are provided here for prosperity.

		LIFE-Family/N (Num. scoring taxa)			
Station	**Status**	**Spring 16**	**Summer 16**	**Spring 17**	**Summer 17**
25A030100	Maintained	7.84 (19)	8.1 (10)	8.47 (15)	8.17 (18)
25B070200	Lost	7.56 (16)	7.67 (12)	7.94 (17)	8.13 (15)
25B100100	Gained	7.59 (22)	7.69 (16)	7.73 (22)	7.94 (16)
25B100200	Maintained	7.71 (21)	7.56 (16)	7.73 (22)	7.71 (14)
25B150050	Lost	7.3 (20)	7.36 (14)	7.61 (18)	7.53 (15)
25B150500	Maintained	7.39 (23)		8 (14)	7.47 (17)
25C030100	Gained	7.53 (17)	7.78 (9)	7.65 (23)	7.92 (13)
25C070200	Gained	7.73 (22)	7.57 (14)	7.62 (21)	7.78 (18)
25C090100	Lost	7.94 (16)	7.81 (16)	7.65 (23)	7.75 (20)
25C090400	Maintained	7.58 (19)	7.81 (16)	7.96 (23)	8 (17)
25D100200	Lost	7.58 (19)	7.67 (15)	8.29 (17)	7.53 (15)
25G040025	Maintained	7.42 (19)	7.43 (21)	8 (19)	7.89 (18)
25W010300	Lost	7.27 (26)		7.82 (22)	7.28 (18)
26D070700	Lost	7.33 (18)	7.33 (15)	7.43 (14)	7.6 (15)
26F020080	Gained	7.18 (17)	7.38 (16)	7.2 (15)	7.61 (18)
26F020250	Gained	7.4 (20)	7.59 (17)	7.71 (14)	7.57 (21)
26I030300	Maintained	7.06 (17)	7.42 (19)	7.38 (13)	7.64 (14)
26I030400	Lost	7.47 (15)	7.93 (15)	7.88 (16)	7.53 (17)
26K010300	Gained	8 (13)	7.36 (28)	7.54 (24)	7.58 (31)
26L030350	Gained	7.45 (20)	7.5 (26)	7.38 (21)	7.54 (26)
26S030400	Maintained	6.88 (17)	6.78 (18)	7.72 (18)	
26Y010200	Gained	7.47 (15)	7 (10)	7.85 (13)	7.83 (18)
27B020600	Lost	7.52 (21)	7.63 (8)	8.31 (13)	7.36 (11)
27G020600	Maintained	7.57 (21)	7.5 (20)	7.81 (21)	7.73 (15)

27O010700	Maintained	7.55 (20)	7.76 (17)	7.5 (18)	
29B020100	Lost	7.26 (23)	7.43 (21)	7.33 (21)	7.29 (24)
29B040300	Maintained	7.55 (20)	7.72 (18)	8.36 (14)	8.38 (13)
29O011000	Maintained	8.18 (17)	7.82 (17)	8.11 (19)	8.18 (17)
30G010250	Maintained	7.5 (12)	7.43 (14)	8.06 (16)	8.13 (16)
30N010100	Lost	7.46 (24)	7.94 (17)	7.68 (19)	7.71 (21)
31R010100	Lost	7.36 (11)		8.4 (5)	8 (8)
32B030050	Maintained	7.88 (16)	8.43 (7)	7.59 (17)	7.55 (11)
32C010020	Maintained	7.23 (13)	8.08 (12)	7.56 (16)	8.15 (13)
32C030150	Gained	7.52 (21)	7.56 (16)	8.2 (15)	7.47 (19)
32C050050	Maintained	7.94 (18)	7.47 (15)	7.8 (15)	7.74 (19)
32E010030	Gained	7.53 (19)	8.08 (12)	7.81 (21)	8.21 (14)
32G070300	Lost	8 (15)	7.6 (15)	7.73 (15)	7.92 (13)
32O040250	Gained	7.47 (17)	7.35 (17)	7.57 (14)	7.29 (7)
33A020100	Lost	7.2 (15)	7.69 (13)	7.88 (8)	7.82 (17)
33B010100	Lost	7.21 (19)	7.56 (18)	7.67 (21)	7.75 (16)
33G020200	Gained	7.87 (15)	7.82 (17)	7.69 (16)	7.57 (14)
33K010200	Maintained	7.81 (21)	7.86 (14)	7.71 (21)	7.88 (16)
33O040050	Gained	7.58 (19)	7.5 (16)	8.05 (19)	8.17 (12)
34C030030	Gained	7.39 (23)	7.41 (22)	7.59 (27)	
34C030150	Gained	7.41 (22)	7.83 (18)	7.78 (23)	7.33 (27)
34C050030	Gained	7.76 (21)	7.81 (16)	7.68 (19)	7.83 (12)
34C100300	Lost	6.46 (13)	7.17 (12)	7.53 (15)	6.88 (16)
34D030800	Lost	7.56 (18)	7.17 (24)	7.39 (23)	7.5 (26)
34G010020	Gained	7.25 (6)	7.54 (13)	7.68 (19)	7.75 (12)
34G020200	Lost	7.47 (17)	7.82 (11)	7.72 (18)	7.75 (12)
34M020100	Gained	7.42 (19)	7.71 (17)	8 (16)	7.86 (14)
34O030200	Maintained	7.52 (21)	7.55 (22)	7.55 (22)	7.67 (24)
34S030050	Gained	7.11 (18)	7.59 (17)	7.65 (20)	7.59 (17)
34T010500	Gained	7.67 (21)	7.72 (18)	7.86 (21)	7.94 (18)
34Y010100	Maintained	7.65 (20)	7.72 (18)	7.76 (21)	7.62 (21)
34Y010400	Maintained	7.62 (21)	7.93 (15)	8.1 (20)	7.71 (17)
34Y020275	Lost	6.87 (15)	7.09 (11)	7.56 (18)	7.5 (18)
35C030200	Maintained	7.92 (12)	7.38 (16)	7.5 (14)	7.56 (16)
35D050020	Maintained	8.07 (14)	8.06 (16)	7.56 (18)	7.56 (16)
35D161000	Lost	7.5 (20)	7.83 (12)	8.27 (15)	7.71 (17)
35F010100	Lost	7.5 (20)	7.65 (17)	7.39 (23)	7.31 (13)
35G020200	Maintained	7.56 (18)	7.4 (15)	7.65 (20)	7.9 (20)
35G030100	Gained	7.6 (20)	7.67 (18)	7.44 (18)	8 (16)
35G040080	Maintained	7.56 (16)	8.14 (14)	7.9 (21)	8 (16)
36R020200	Maintained	7.44 (16)	7.18 (17)	7.45 (20)	7.47 (17)

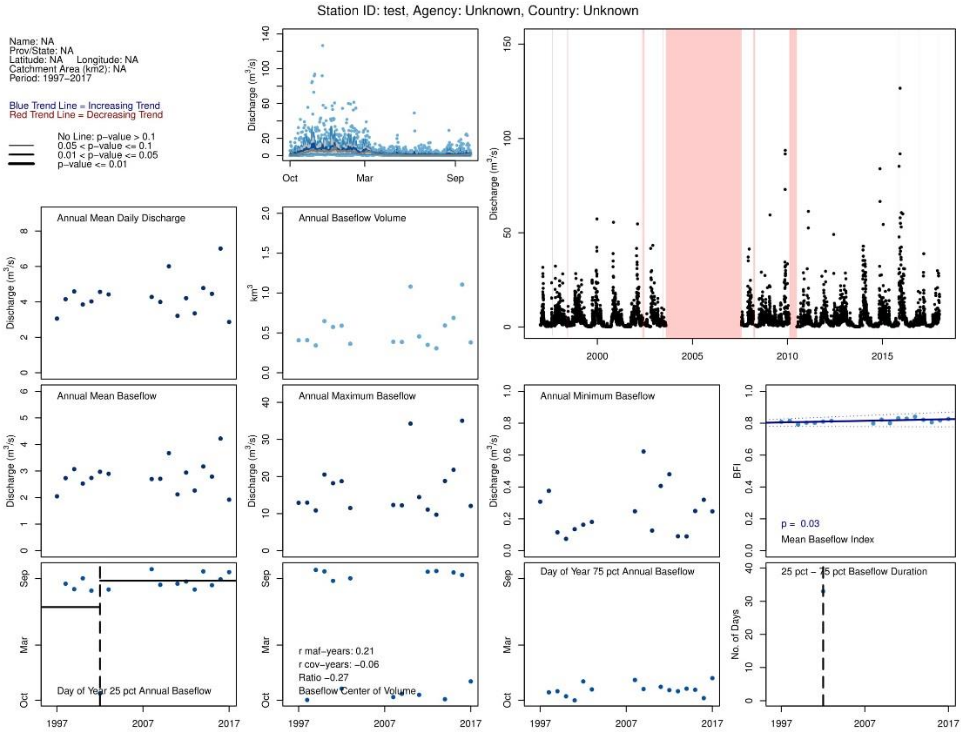


Figure C.1. Output from the R package "Flowscreen" summarising the daily streamflow time series data from the hydrometric station 25020, and identifying significant trends.

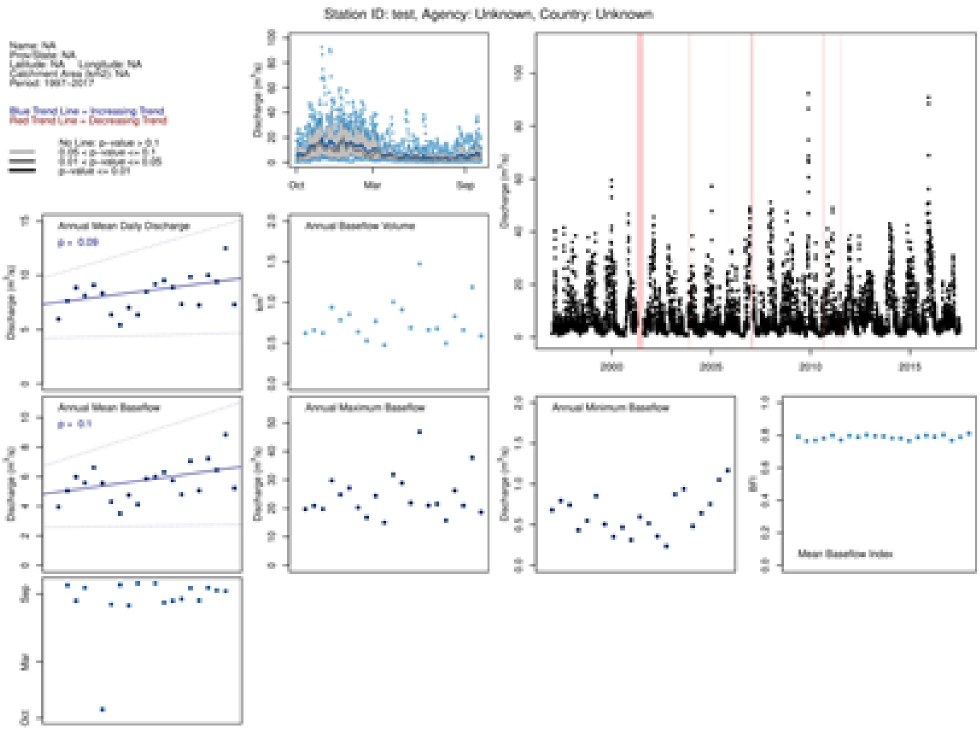


Figure C.2. Output from the R package "Flowscreen" summarising the daily streamflow time series data from the hydrometric station 25030, and identifying significant trends.

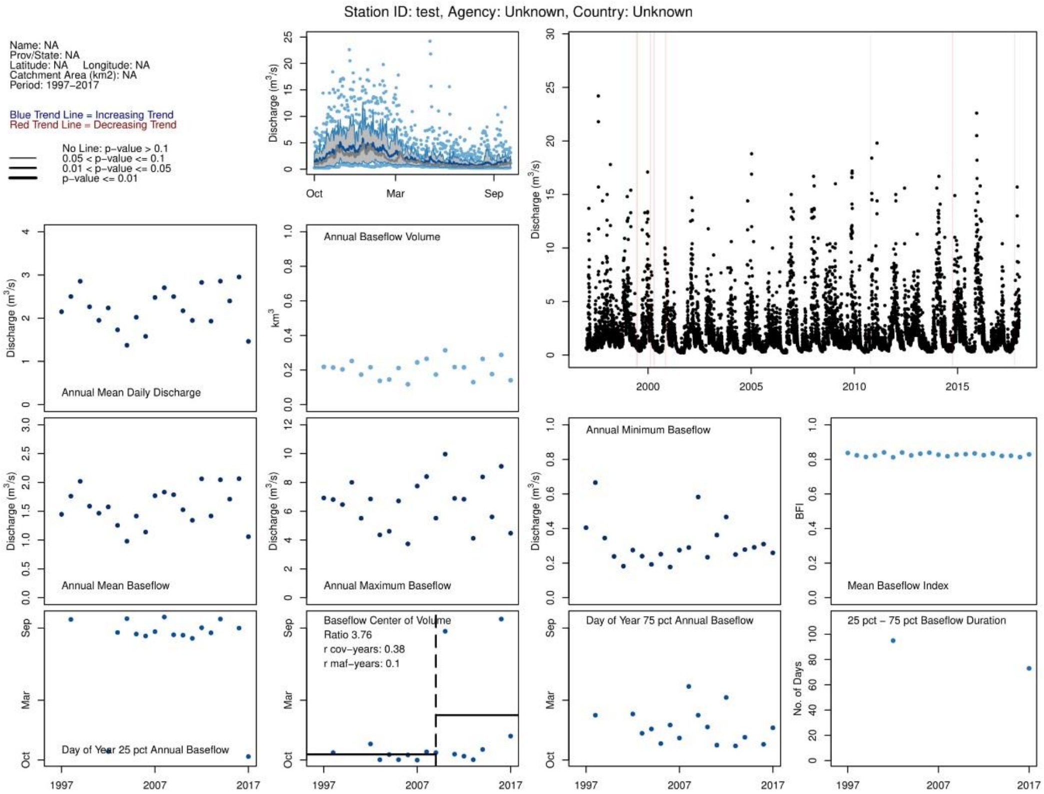


Figure C.3. Output from the R package "Flowscreen" summarising the daily streamflow time series data from the hydrometric station 25044, and identifying significant trends.

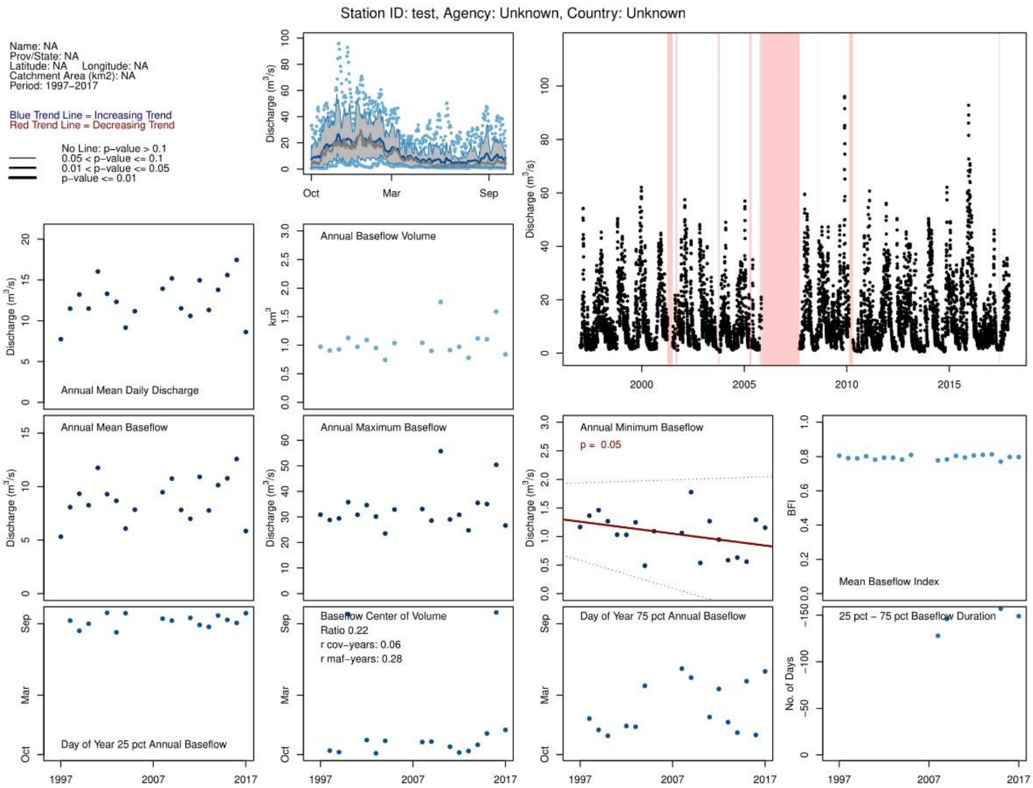


Figure C.4. Output from the R package "Flowscreen" summarising the daily streamflow time series data from the hydrometric station 26002, and identifying significant trends.

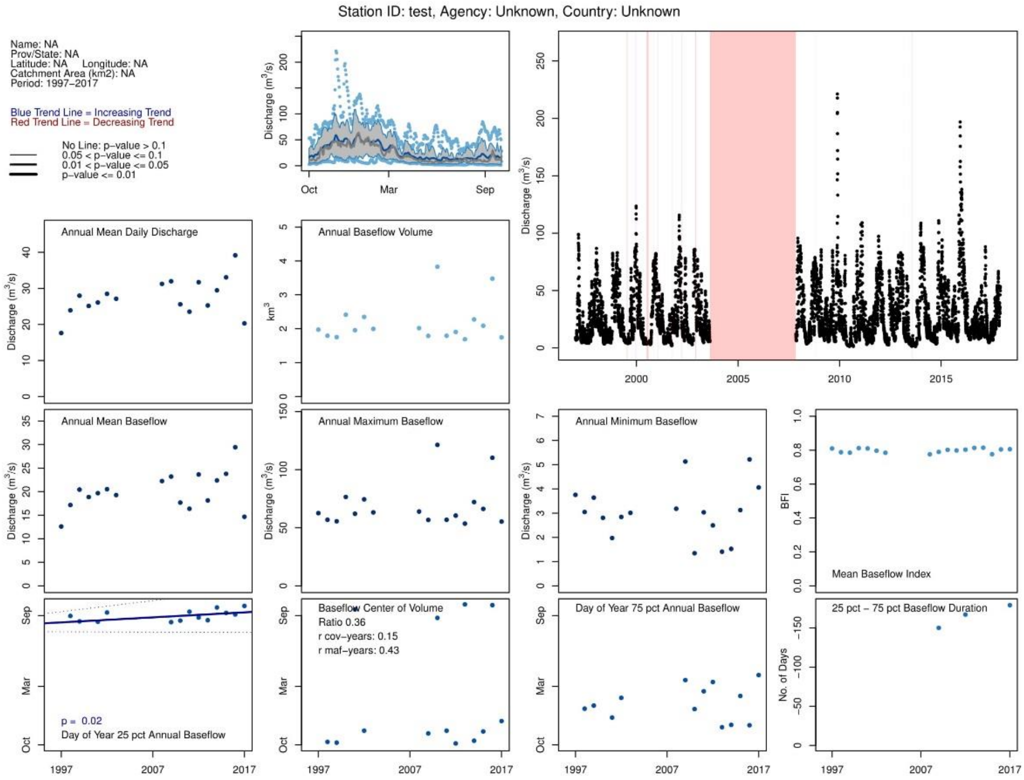


Figure C.5. Output from the R package “Flowscreen” summarising the daily streamflow time series data from the hydrometric station 26007, and identifying significant trends.

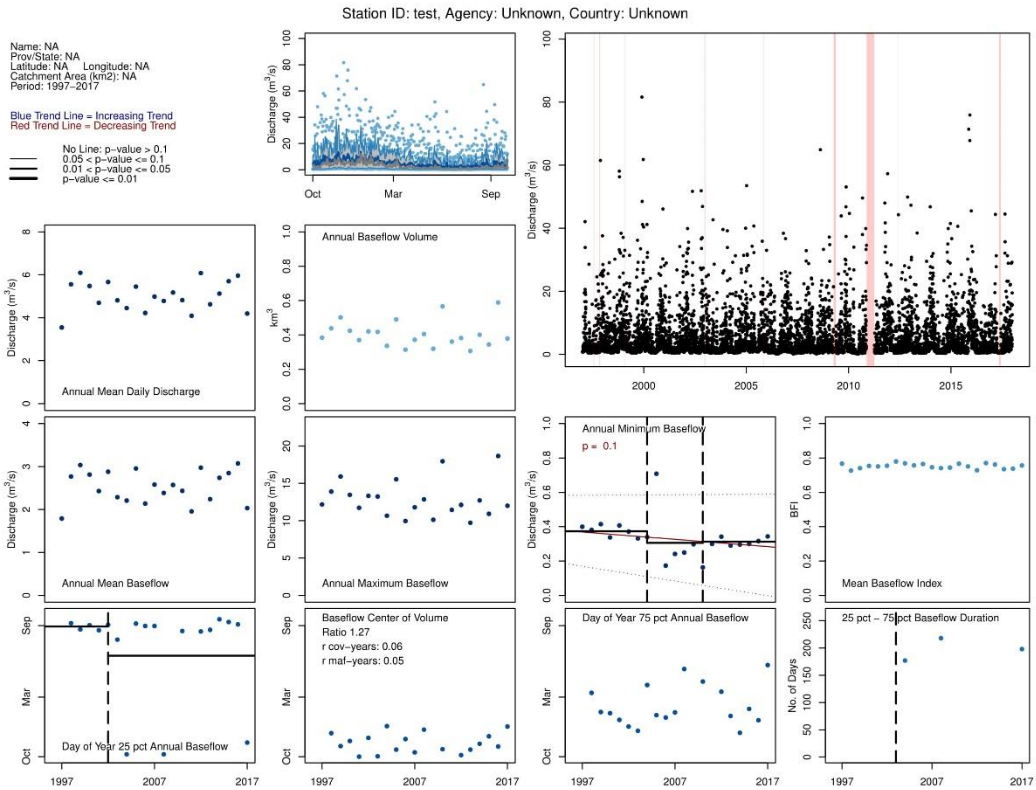


Figure C.6. Output from the R package “Flowscreen” summarising the daily streamflow time series data from the hydrometric station 26029, and identifying significant trends.

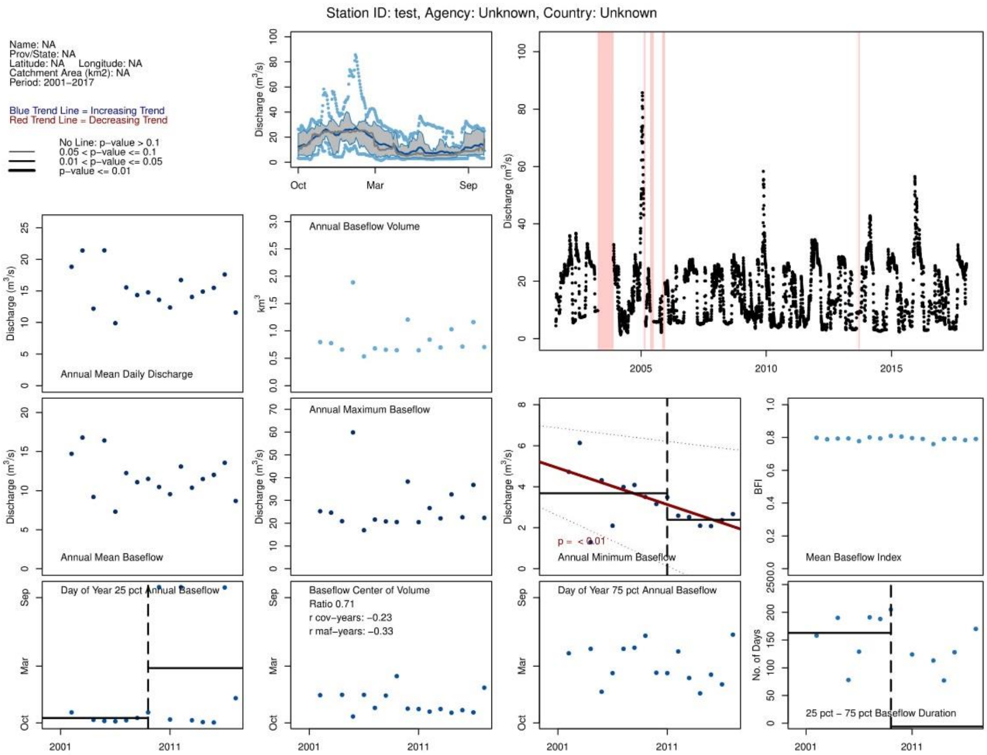


Figure C.7. Output from the R package "Flowscreen" summarising the daily streamflow time series data from the hydrometric station 26030, and identifying significant trends.

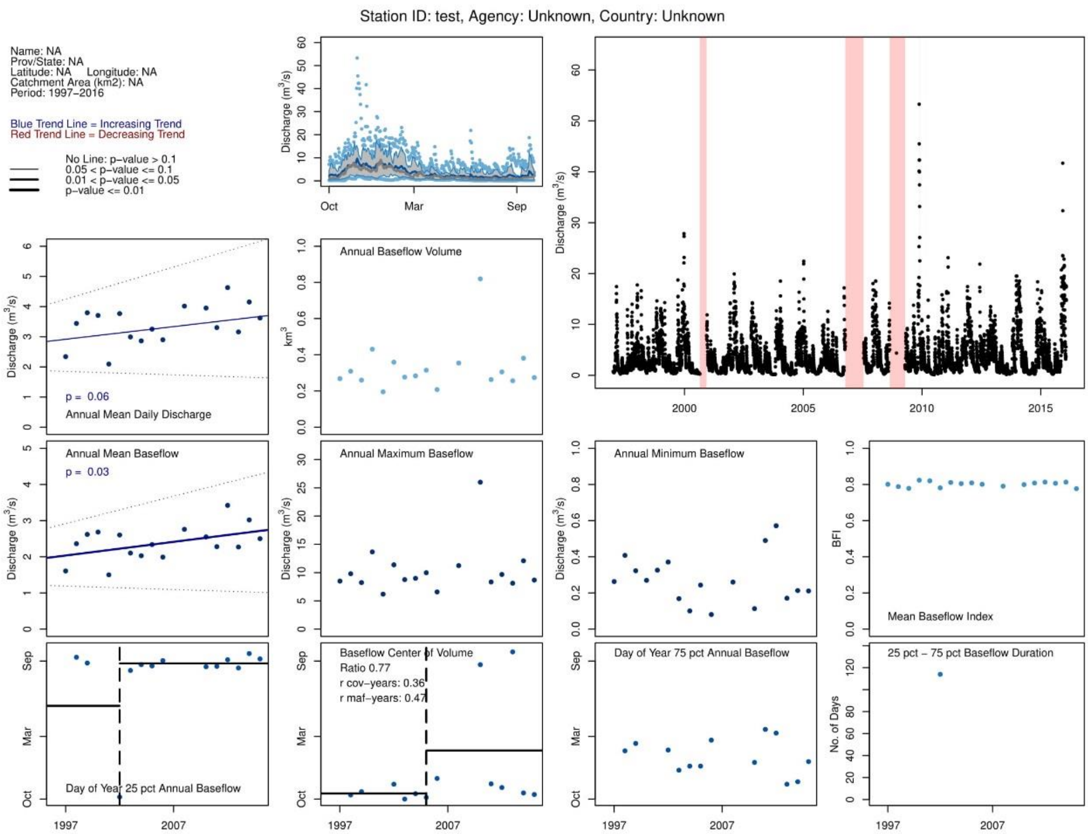


Figure C.8. Output from the R package "Flowscreen" summarising the daily streamflow time series data from the hydrometric station 29071, and identifying significant trends.

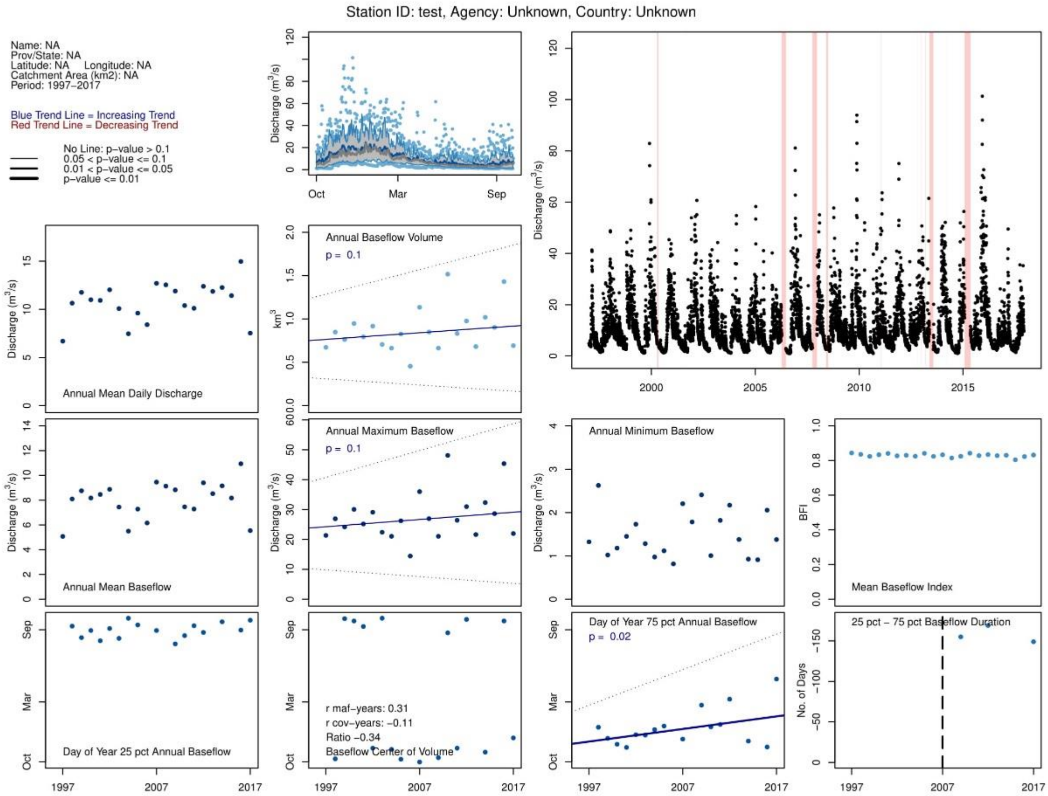


Figure C.9. Output from the R package "Flowscreen" summarising the daily streamflow time series data from the hydrometric station 30007, and identifying significant trends.

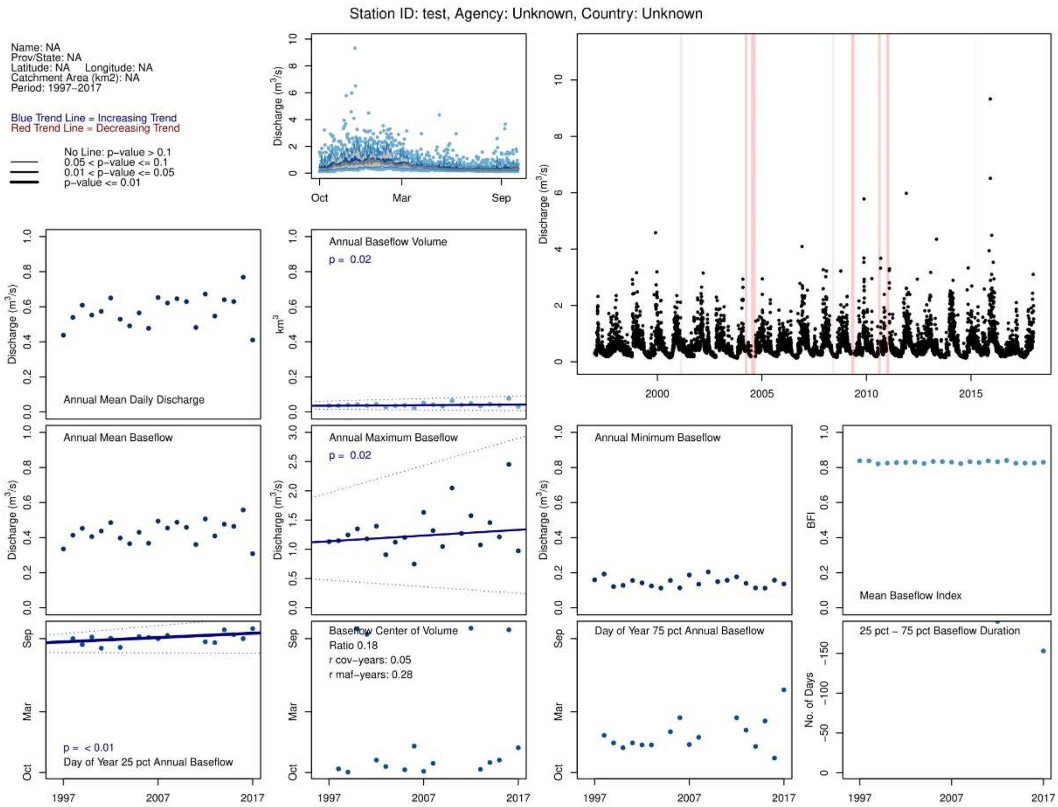


Figure C.10. Output from the R package "Flowscreen" summarising the daily streamflow time series data from the hydrometric station 30020, and identifying significant trends.

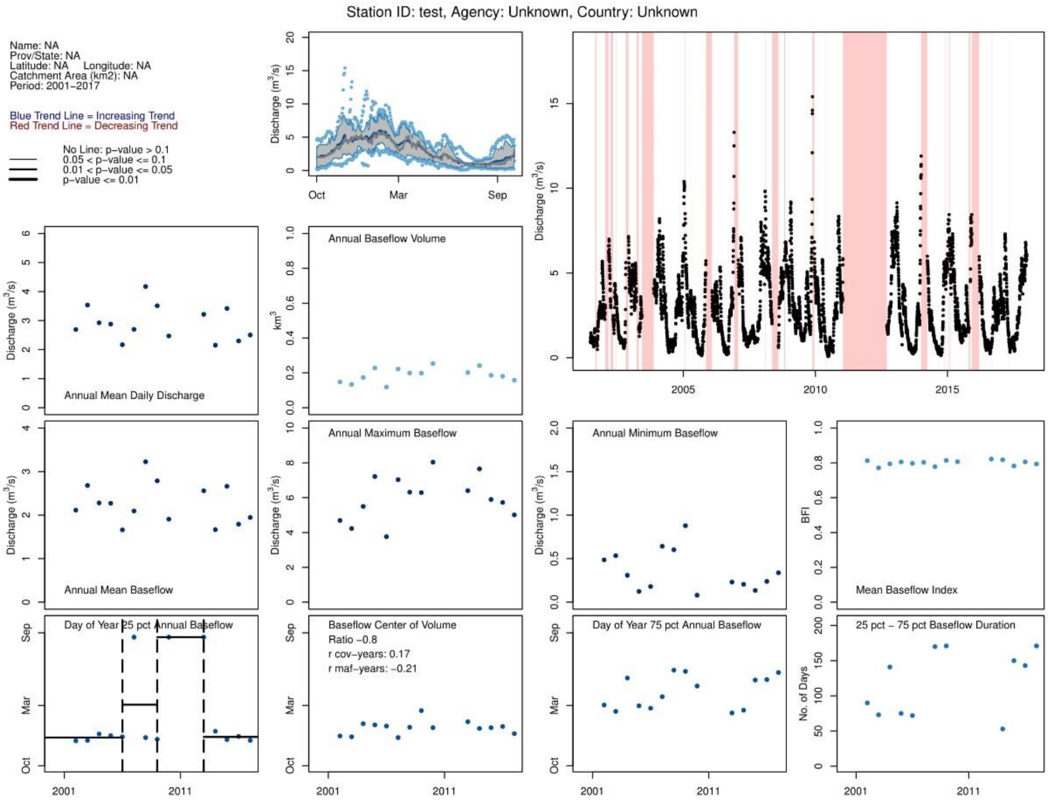


Figure C.11. Output from the R package "Flowscreen" summarising the daily streamflow time series data from the hydrometric station 30047, and identifying significant trends.

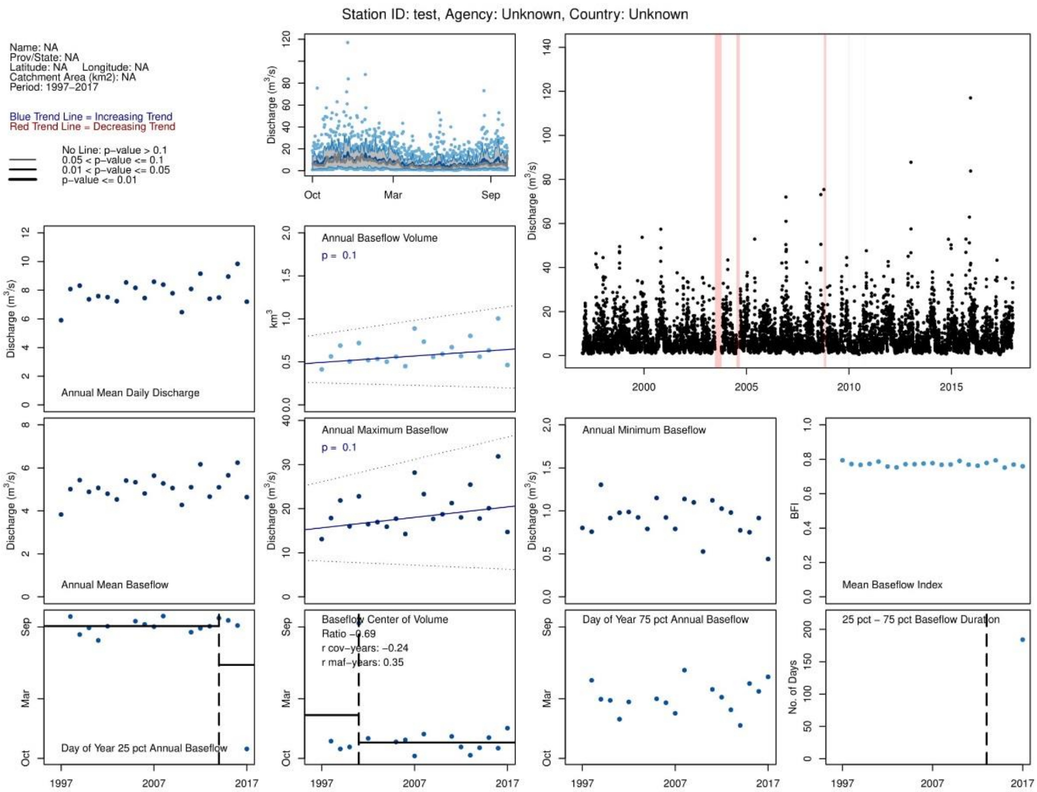


Figure C.12. Output from the R package "Flowscreen" summarising the daily streamflow time series data from the hydrometric station 31072, and identifying significant trends.

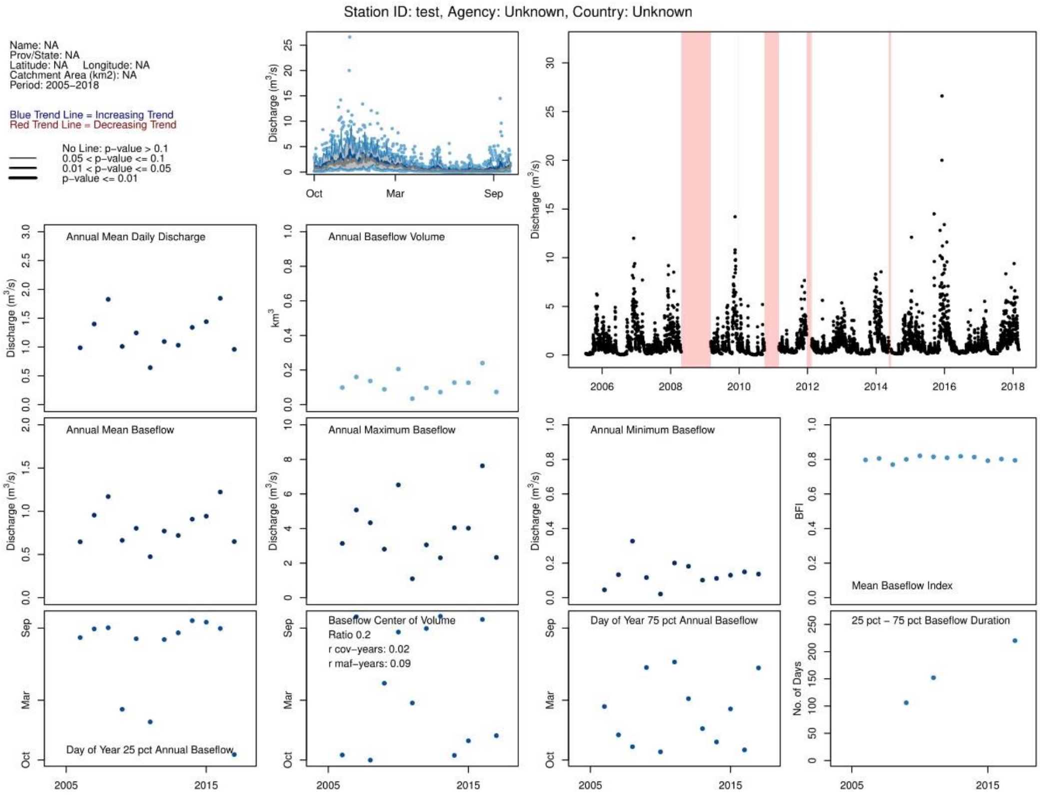


Figure C.13. Output from the R package "Flowscreen" summarising the daily streamflow time series data from the hydrometric station 32006, and identifying significant trends.

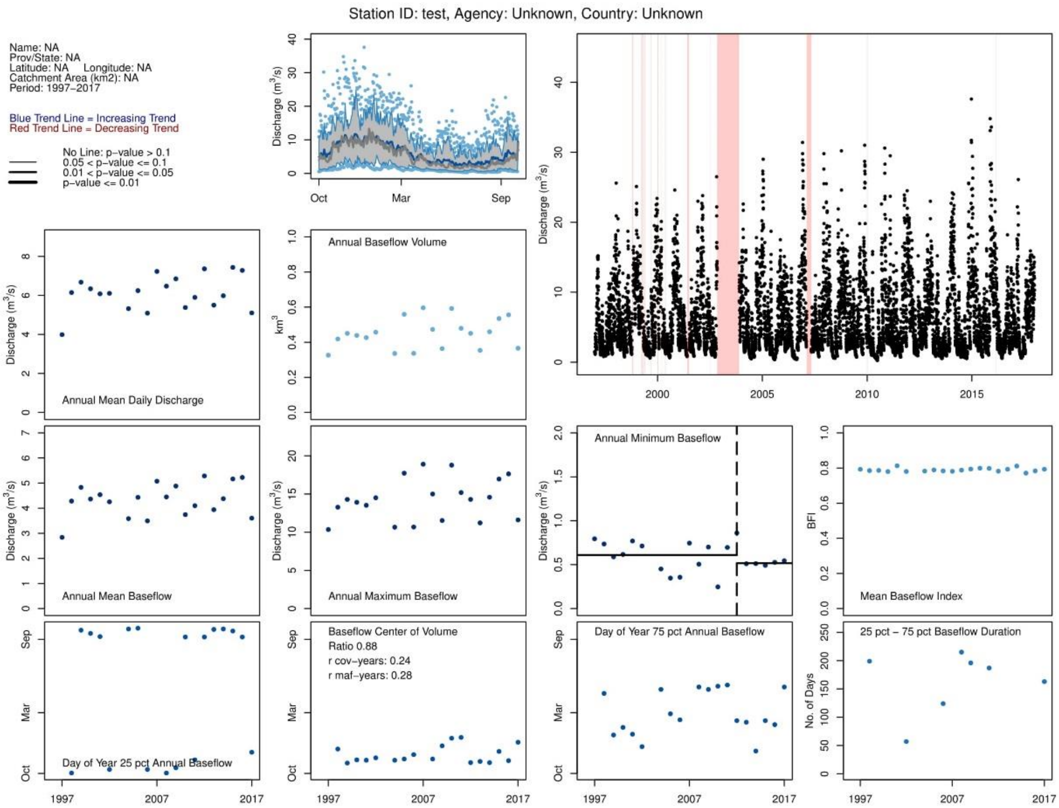


Figure C.14. Output from the R package "Flowscreen" summarising the daily streamflow time series data from the hydrometric station 32012, and identifying significant trends.

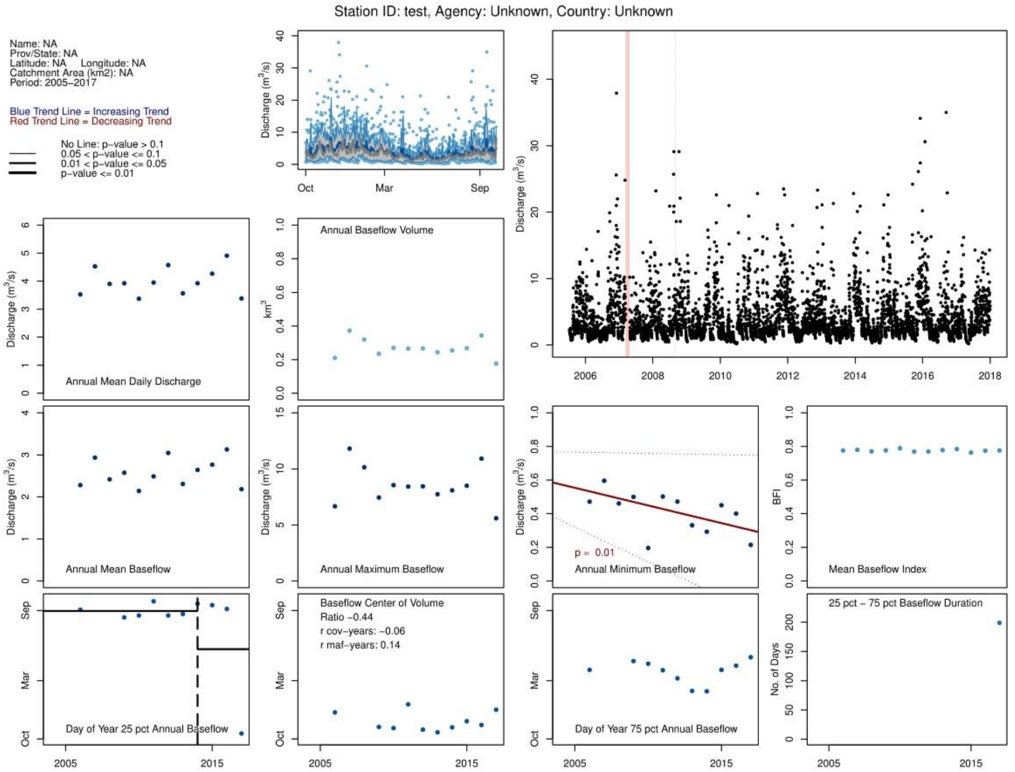


Figure C.15. Output from the R package "Flowscreen" summarising the daily streamflow time series data from the hydrometric station 32026, and identifying significant trends.

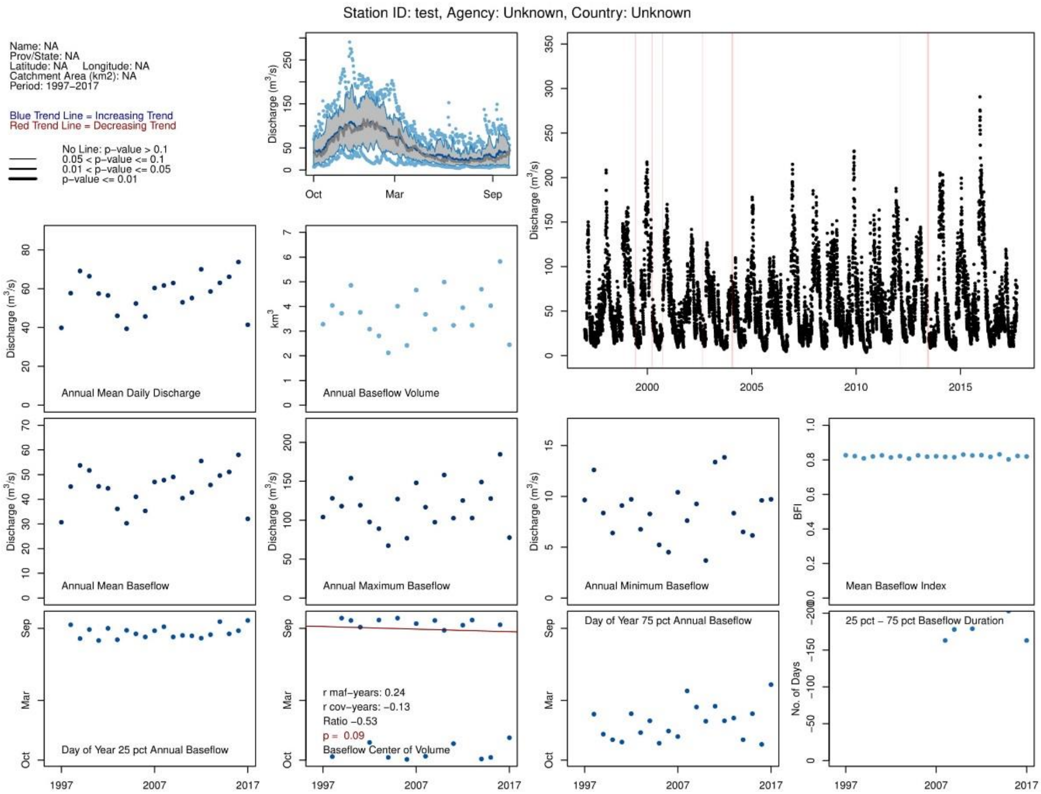


Figure C.16. Output from the R package "Flowscreen" summarising the daily streamflow time series data from the hydrometric station 34001, and identifying significant trends.

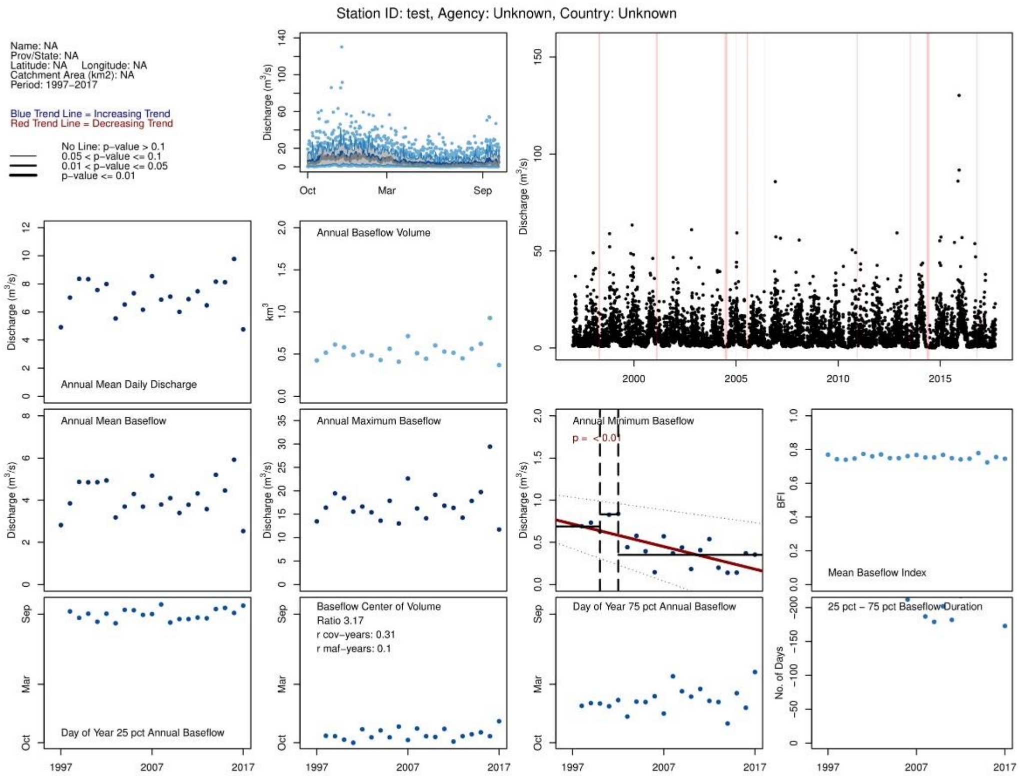


Figure C.17. Output from the R package "Flowscreen" summarising the daily streamflow time series data from the hydrometric station 34007, and identifying significant trends.

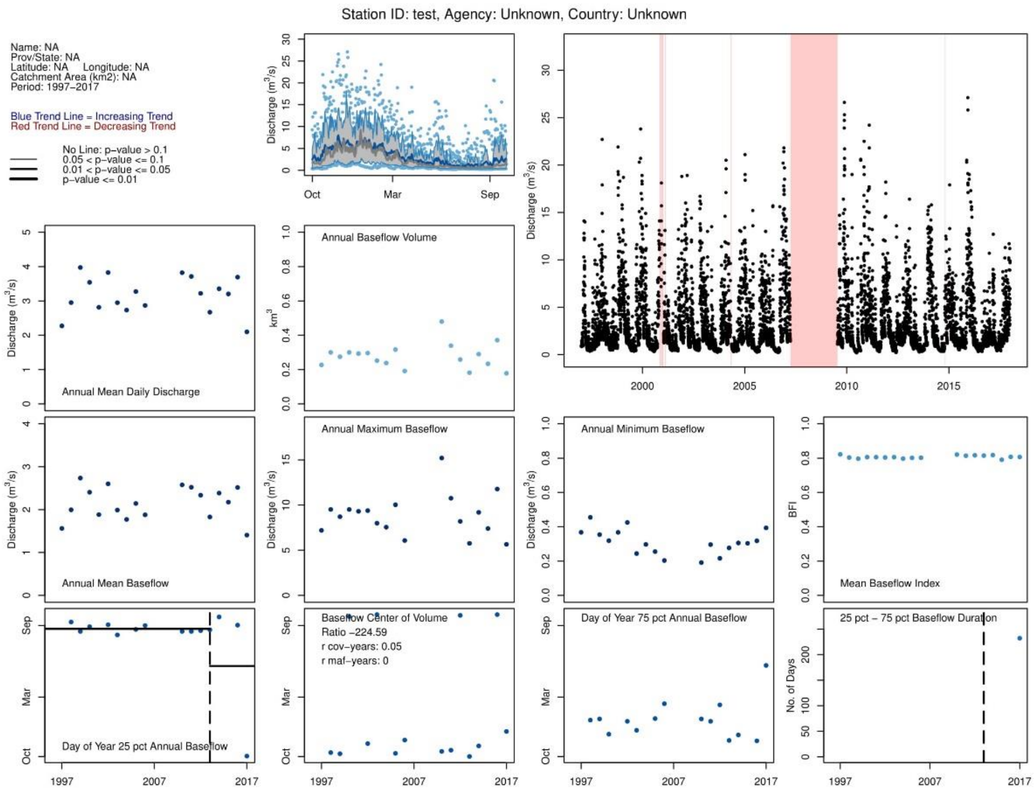


Figure C.18. Output from the R package "Flowscreen" summarising the daily streamflow time series data from the hydrometric station 34024, and identifying significant trends.

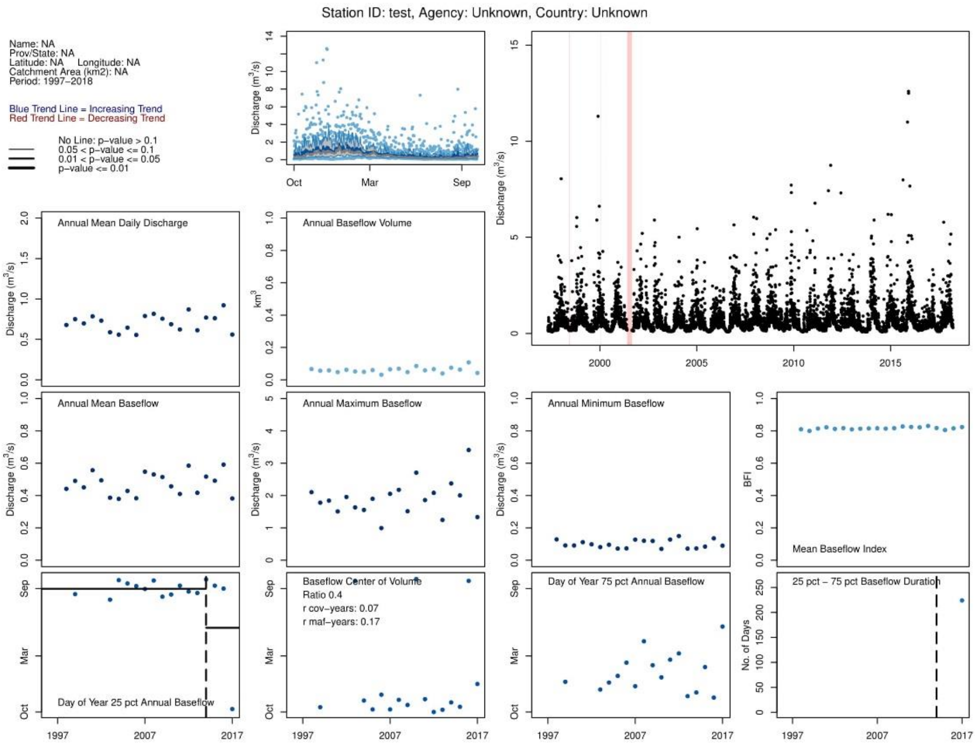


Figure C.19. Output from the R package "Flowscreen" summarising the daily streamflow time series data from the hydrometric station 34031, and identifying significant trends.

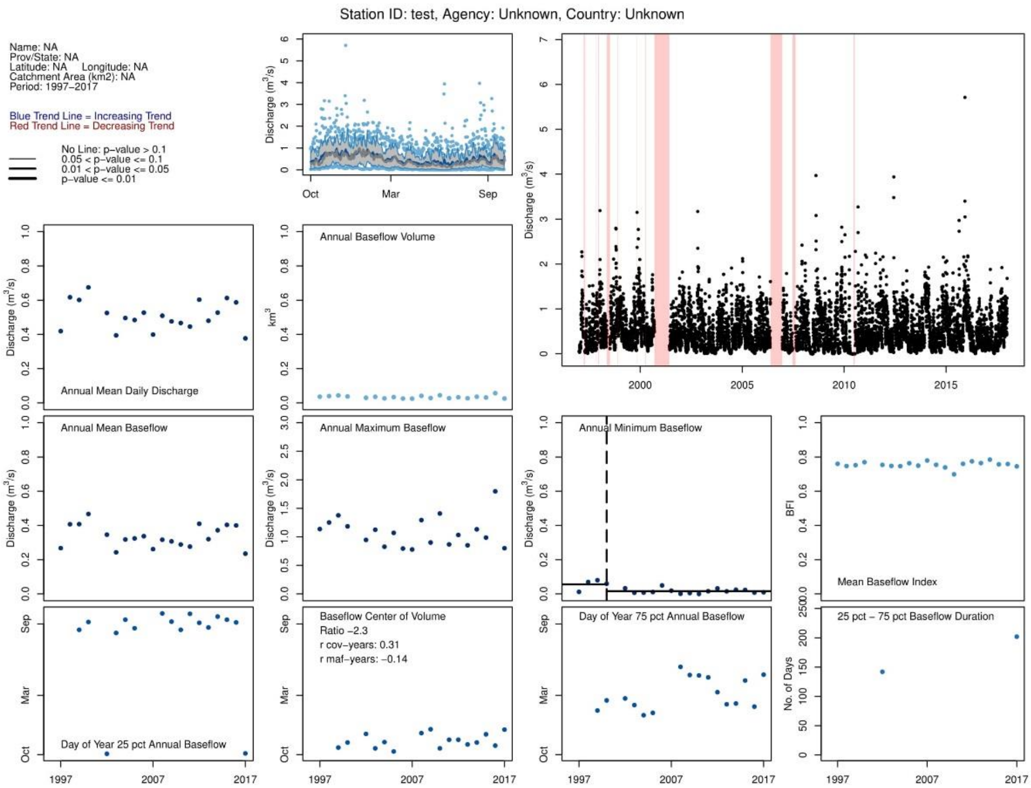
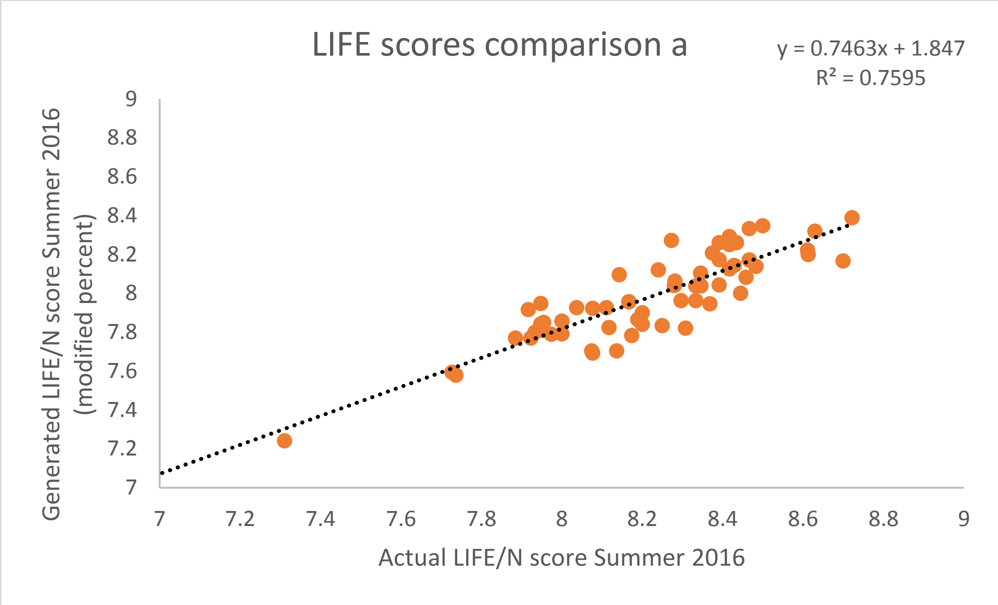
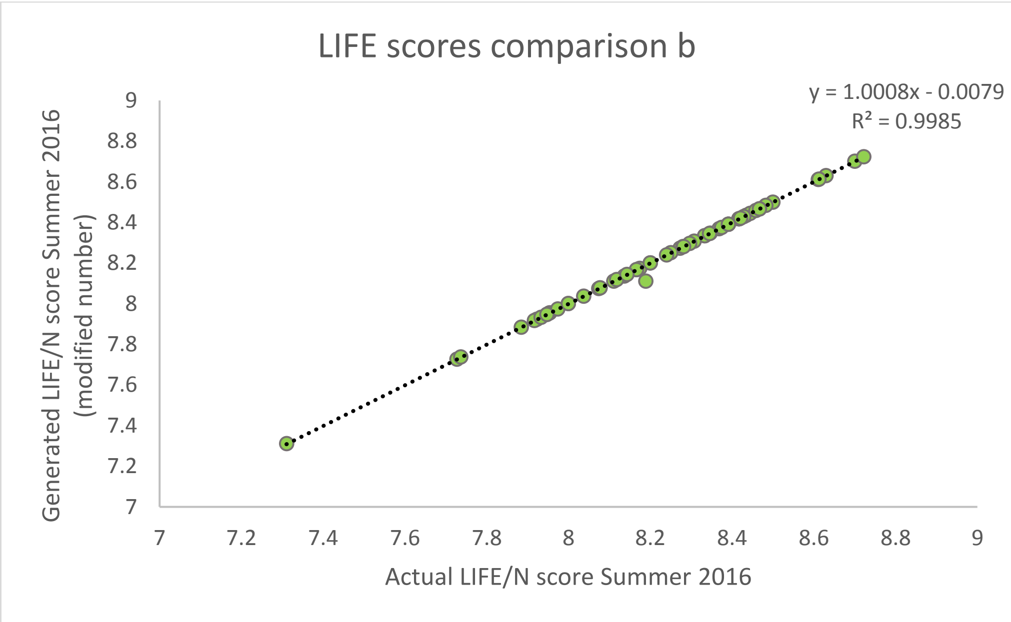


Figure C.20. Output from the R package "Flowscreen" summarising the daily streamflow time series data from the hydrometric station 35072, and identifying significant trends.



a)

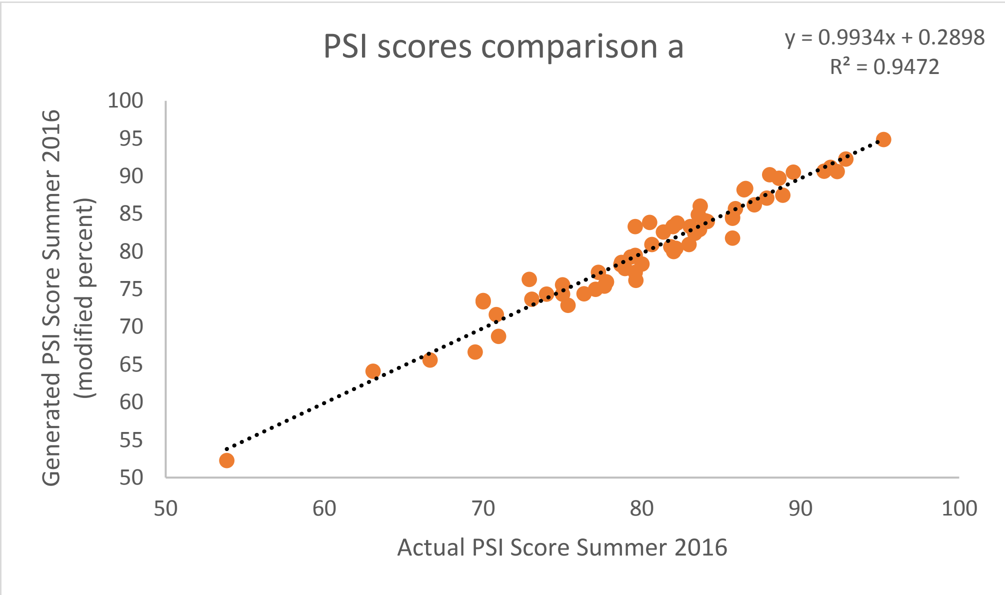


b)

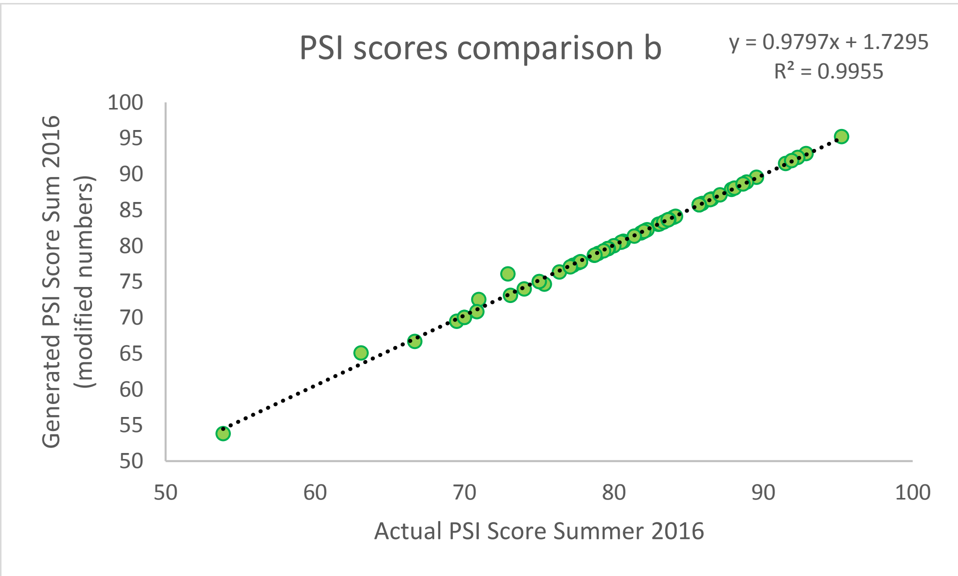
Figure C21. A correlation of the actual LIFE scores from Summer 2016 against the generated LIFE scores from Summer 2016 using either: a) percentage abundances assigned as either Single, Few, Common etc.; or b) directly assigning the categories Few, Common, etc. based on the numbers of taxa present, without first calculating the abundance percentages.

Appendix D

Supplementary data relating to Chapter 4 - Assessing the impact of fine sediment on high status river sites in Ireland



a)



b)

Figure D1. A correlation of the actual PSI scores from Summer 2016 against the generated PSI scores from Summer 2016 using either: a) percentage abundances assigned as either Single, Few, Common etc.; or b) directly assigning the categories Few, Common, etc. based on the numbers of taxa present, without first calculating the abundance percentages.

Table D.1. List of PSI scores [species/mixed taxon level] occurring at the sixty-five sample sites during Spring 2016, Summer 2016, Spring 2017 and Summer 2017.

		PSI scores			
Station	**Status**	**Spring 16**	**Summer 16**	**Spring 17**	**Summer 17**
25A030100	Maintained	95.71	91.89	92.73	90.24
25B070200	Lost	83.72	88.89	78.18	92.16
25B100100	Gained	83.82	88.64	89.04	91.18
25B100200	Maintained	87.67	85.71	84.93	81.25
25B150050	Lost	77.36	79.59	82.35	84.21
25B150500	Maintained	75.76		91.53	61.90
25C030100	Gained	80.70	77.78	87.32	80
25C070200	Gained	88.57	82.22	83.33	83.33
25C090100	Lost	91.53	85.71	86.76	88.46
25C090400	Maintained	88.71	82.98	84.62	87.27
25D100200	Lost	84.09	78.95	91.07	83.33
25G040025	Maintained	82.35	81.97	87.10	81.63
25W010300	Lost	72.37		79.17	64.29
26D070700	Lost	77.55	73.08	71.05	73.91
26F020080	Gained	69.57	70	66.67	74.14
26F020250	Gained	78.95	75	82.98	76.74
26I030300	Maintained	75.00	69.49	71.05	67.57
26I030400	Lost	76.60	80.65	85.45	64.29
26K010300	Gained	80.95	70.97	75.00	69.41
26L030350	Gained	75.86	75.34	72.88	76.47
26S030400	Maintained	55	53.85	79.63	
26Y010200	Gained	83.72	77.27	87.88	83.72
27B020600	Lost	81.97	77.78	93.02	70
27G020600	Maintained	89.86	78.72	89.55	85
27O010700	Maintained	75.51	77.08	73.08	
29B020100	Lost	62.69	63.08	62.96	63.33
29B040300	Maintained	89.83	86.44	87.76	89.74
29O011000	Maintained	89.66	84.13	87.72	87.93
30G010250	Maintained	76.92	87.88	92	87.23
30N010100	Lost	70.37	72.92	75.44	66.67
31R010100	Lost	85.19		76.47	95
32B030050	Maintained	90.91	85.71	78.57	76.19
32C010020	Maintained	85.29	91.49	84.62	82.35
32C030150	Gained	86.89	86.54	90.74	75
32C050050	Maintained	87.50	81.82	88	80.77
32E010030	Gained	82.69	75	83.33	88.10
32G070300	Lost	85.71	79.59	80.95	82.35
32O040250	Gained	76.32	74	78.57	63.16
33A020100	Lost	75.61	76.36	85.71	80.39
33B010100	Lost	77.19	78.69	84.85	76.19
33G020200	Gained	88.89	82.14	77.78	89.66
33K010200	Maintained	85.48	95.24	85.96	86.84
33O040050	Gained	88.24	79.59	84.21	78.05
34C030030	Gained	82.35	85.90	86.96	
34C030150	Gained	76.47	83.56	81.08	72.22
34C050030	Gained	87.27	84.00	82.54	87.50
34C100300	Lost	34.48	66.67	78.72	56.67
34D030800	Lost	82.69	77.65	82.09	75.00
34G010020	Gained	86.67	79.31	80.85	80
34G020200	Lost	83.33	87.10	86.05	77.42

34M020100	Gained	83.33	89.55	89.47	86.96
34O030200	Maintained	84.85	70.83	79.03	79.71
34S030050	Gained	80.77	83.05	81.97	79.25
34T010500	Gained	85.48	81.36	81.69	86.79
34Y010100	Maintained	83.87	83.33	90.28	78.00
34Y010400	Maintained	86.76	83.67	90.41	76.92
34Y020275	Lost	64.10	70	74	67.35
35C030200	Maintained	88.89	82	84.09	86.00
35D050020	Maintained	93.75	92.86	86.89	88.24
35D161000	Lost	83.64	80.49	88.89	74.42
35F010100	Lost	90.91	83.64	82.81	76
35G020200	Maintained	83.64	80.00	88.33	88.24
35G030100	Gained	88.46	88.06	87.72	78.95
35G040080	Maintained	91.07	92.31	87.10	88.64
36R020200	Maintained	80.77	79.63	79.63	78.26

Table D.2. List of Co-FSI scores for the sixty-five sample sites during Spring 2016, Summer 2016, Spring 2017 and Summer 2017.

		CoFSI Scores			
Station	**Status**	**Spring 16**	**Summer 16**	**Spring 17**	**Summer 17**
25A030100	Maintained	151.02	98.47	123.92	88.55
25B070200	Lost	98.48	100.78	105.65	99.08
25B100100	Gained	157.92	112.85	171.12	93.89
25B100200	Maintained	159.81	114.08	172.02	96.41
25B150050	Lost	146.68	115.38	148.03	114.37
25B150500	Maintained	154.49		123.11	65.33
25C030100	Gained	143.06	78.97	161.54	71.2
25C070200	Gained	154.47	99.33	133.79	111.8
25C090100	Lost	131.48	102.05	153.24	115.6
25C090400	Maintained	145.03	109.39	170.15	107.56
25D100200	Lost	107.78	98.99	135.36	101.09
25G040025	Maintained	140.88	145.2	136.68	125.14
25W010300	Lost	160.39		171.4	122.32
26D070700	Lost	102.53	122.72	101	92.56
26F020080	Gained	108.64	81.29	118.3	131.65
26F020250	Gained	122.58	104.79	111.67	107.85
26I030300	Maintained	147.59	121.46	120.58	81.07
26I030400	Lost	114.81	125.02	130.36	106.2
26K010300	Gained	102.49	156.64	159.08	156.27
26L030350	Gained	128.14	140.41	135.12	140.5
26S030400	Maintained	111.18	121.24	125.68	
26Y010200	Gained	118.1	50.13	107.48	102.36
27B020600	Lost	131.96	62.15	95.99	65.64
27G020600	Maintained	158.66	120.05	159.17	95.09
27O010700	Maintained	112.15	117.84	130.16	
29B020100	Lost	127.87	124.07	135.82	123.72
29B040300	Maintained	140.86	121.15	121.68	84.48
29O011000	Maintained	119.18	110.96	138.9	111.49
30G010250	Maintained	88.77	92.56	126.75	107.25
30N010100	Lost	128.16	78.83	141.64	114.32
31R010100	Lost	62.9		46.6	59.4
32B030050	Maintained	115.49	62.15	132	52.88
32C010020	Maintained	110.29	103.67	130.91	75.13
32C030150	Gained	169.14	120.36	132.12	108.07
32C050050	Maintained	135.54	115.03	114.96	114.81
32E010030	Gained	137.54	88.06	168.39	109.65
32G070300	Lost	109.98	134.04	138.9	90.69
32O040250	Gained	89.9	107.91	112.1	46.42
33A020100	Lost	118.94	116.65	52.13	117.22
33B010100	Lost	131.78	120.19	156.55	91.73
33G020200	Gained	105.78	116.59	136.24	79.45
33K010200	Maintained	141.27	106.42	149.53	110.88
33O040050	Gained	138.02	125.11	141.86	94.62
34C030030	Gained	159.52	141.99	177.44	

34C030150	Gained	152.89	148.19	184.88	167.42
34C050030	Gained	114.83	121.2	132.31	73.61
34C100300	Lost	66.47	95.79	127.44	90.28
34D030800	Lost	115.38	160.73	182.2	154.84
34G010020	Gained	102.01	82.64	134.66	71.72
34G020200	Lost	97.07	75.87	110.66	70.45
34M020100	Gained	149.22	136.62	126.71	98.38
34O030200	Maintained	135.08	153.65	157.54	148.5
34S030050	Gained	119.97	110.92	147.42	112.32
34T010500	Gained	148.69	126.09	170.13	137.1
34Y010100	Maintained	134.71	136.46	185.54	110.88
34Y010400	Maintained	152.19	112.22	185.1	118.92
34Y020275	Lost	98.77	67	134.11	115.51
35C030200	Maintained	109.83	102.05	124.61	111.19
35D050020	Maintained	102.19	106.38	140.62	96.25
35D161000	Lost	140.92	89.46	101.87	86.22
35F010100	Lost	117.97	115.42	150.92	88.72
35G020200	Maintained	122.39	115.56	163.28	100.53
35G030100	Gained	145.58	131.68	155.77	79.32
35G040080	Maintained	123.4	97.04	153.99	99.04
36R020200	Maintained	112.76	115.16	137.01	84.7

Table D.3. List of E-PSI (%) scores for the sixty-five sample sites during Spring 2016, Summer 2016, Spring 2017 and Summer 2017.

		E-PSI (%)			
Station	**Status**	**Spring 16**	**Summer 16**	**Spring 17**	**Summer 17**
25A030100	Maintained	99.63	100	98.26	97.01
25B070200	Lost	96.97	98.36	95.22	98.38
25B100100	Gained	95.5	97.96	98.88	99.02
25B100200	Maintained	97.67	95.47	98.09	97.23
25B150050	Lost	93.64	93.02	97.78	96.59
25B150500	Maintained	93.24		98.71	80.92
25C030100	Gained	94.49	95.1	95.94	95.83
25C070200	Gained	96.91	96.11	94.98	95.93
25C090100	Lost	100	95.62	98.02	96.63
25C090400	Maintained	98.19	95.38	95.91	97.27
25D100200	Lost	95.32	94.54	98.48	95.38
25G040025	Maintained	96.27	94.4	99.55	95.13
25W010300	Lost	89.38		92.44	86.45
26D070700	Lost	89.73	89.71	92.74	89.78
26F020080	Gained	85.28	87.8	86.5	91.61
26F020250	Gained	87.65	88.93	95.53	87.7
26I030300	Maintained	87.28	85.33	88.8	80.3
26I030400	Lost	93.62	95.93	96.32	84.89
26K010300	Gained	97.05	84.42	89.72	83.89
26L030350	Gained	91.82	88	88.74	86.02
26S030400	Maintained	80.04	72.44	95.62	
26Y010200	Gained	97.62	91.26	100	96.56
27B020600	Lost	95.84	97.54	100	91.98
27G020600	Maintained	100	97.3	100	97.63
27O010700	Maintained	89.16	92.26	87.7	
29B020100	Lost	85.07	86.18	84.52	84.11
29B040300	Maintained	99.57	97.42	100	100
29O011000	Maintained	99.53	96.68	98.04	97.93
30G010250	Maintained	98.96	99.14	100	97.83
30N010100	Lost	87.09	86.27	91.52	79.6
31R010100	Lost	98.36		98.5	98.96
32B030050	Maintained	100	100	99.38	100
32C010020	Maintained	100	100	99.01	95.69
32C030150	Gained	96.44	98.04	100	93.17
32C050050	Maintained	97.6	96.77	100	95.65
32E010030	Gained	97.2	96.28	97.92	99.3
32G070300	Lost	100	93.72	93.16	94.66
32O040250	Gained	90.66	87.76	91.47	80.29
33A020100	Lost	99.27	94.79	95.83	96.4
33B010100	Lost	94.95	94.4	95.66	91.1
33G020200	Gained	98.41	96.51	95.36	96.47
33K010200	Maintained	96.85	100	98.67	98
33O040050	Gained	99.5	93.46	96.47	96.06
34C030030	Gained	96.59	97.22	95.88	
34C030150	Gained	95.38	94.96	96.29	88.89
34C050030	Gained	99.53	95.57	97.69	100
34C100300	Lost	55.08	80.37	90.09	72.8
34D030800	Lost	94.32	94.52	94.29	93.99
34G010020	Gained	96.98	95.49	94.94	96.25
34G020200	Lost	97.93	100	97.89	97.07

34M020100	Gained	97.75	97.48	100	98.07
34O030200	Maintained	96.25	88.46	91.81	91.35
34S030050	Gained	95.25	93.87	90.88	91.75
34T010500	Gained	95.93	97.42	96.69	97.93
34Y010100	Maintained	96.17	96.1	99.61	95.09
34Y010400	Maintained	98.25	95.89	98.9	92.33
34Y020275	Lost	73.97	74.62	85.32	81.5
35C030200	Maintained	100	95.25	98.08	96.7
35D050020	Maintained	98.02	100	96.69	91.58
35D161000	Lost	92.43	93.22	98	92.78
35F010100	Lost	95.4	96.89	94.98	87.53
35G020200	Maintained	96.32	95.87	98.01	97.39
35G030100	Gained	100	97.9	100	96.89
35G040080	Maintained	100	100	98.81	100
36R020200	Maintained	93.24	95.04	92.79	92.75

Table D.4. List of BMWP, N-taxa and ASPT (BMWP) scores for the sixty-five sample sites during Spring 2016 and Summer 2016.

		Spring 2016			Summer 2016		
Station	**Status**	**BMWP**	**N-taxa**	**ASPT**	**BMWP**	**N-taxa**	**ASPT**
25A030100	Maintained	134	19	7.05	67	11	6.09
25B070200	Lost	88	15	5.87	60	11	5.45
25B100100	Gained	137	21	6.52	103	16	6.44
25B100200	Maintained	146	21	6.95	93	16	5.81
25B150050	Lost	119	19	6.26	76	14	5.43
25B150500	Maintained	110	19	5.79			
25C030100	Gained	97	16	6.06	56	10	5.6
25C070200	Gained	144	21	6.86	81	14	5.79
25C090100	Lost	107	16	6.69	95	15	6.33
25C090400	Maintained	131	19	6.89	105	17	6.18
25D100200	Lost	120	19	6.32	99	16	6.19
25G040025	Maintained	112	18	6.22	122	19	6.42
25W010300	Lost	129	22	5.86			
26D070700	Lost	88	15	5.87	68	13	5.23
26F020080	Gained	93	16	5.81	84	14	6
26F020250	Gained	121	18	6.72	89	15	5.93
26I030300	Maintained	83	15	5.53	95	17	5.59
26I030400	Lost	73	13	5.62	78	14	5.57
26K010300	Gained	78	13	6	137	24	5.71
26L030350	Gained	114	19	6	141	23	6.13
26S030400	Maintained	82	16	5.13	81	17	4.76
26Y010200	Gained	101	16	6.31	60	11	5.45
27B020600	Lost	131	21	6.24	36	8	4.5
27G020600	Maintained	134	20	6.7	118	19	6.21
27O010700	Maintained	97	17	5.71	92	16	5.75
29B020100	Lost	127	21	6.05	103	18	5.72
29B040300	Maintained	137	20	6.85	105	16	6.56
29O011000	Maintained	115	16	7.19	94	16	5.88
30G010250	Maintained	76	13	5.85	72	13	5.54
30N010100	Lost	137	22	6.23	97	16	6.06
31R010100	Lost	59	11	5.36			
32B030050	Maintained	107	16	6.69	46	8	5.75
32C010020	Maintained	87	14	6.21	78	13	6
32C030150	Gained	108	17	6.35	87	14	6.21
32C050050	Maintained	114	17	6.71	81	14	5.79
32E010030	Gained	106	16	6.63	73	13	5.62
32G070300	Lost	99	15	6.6	88	15	5.87
32O040250	Gained	110	17	6.47	91	16	5.69
33A020100	Lost	81	14	5.79	63	12	5.25
33B010100	Lost	92	16	5.75	90	16	5.63
33G020200	Gained	80	13	6.15	90	14	6.43
33K010200	Maintained	133	20	6.65	76	12	6.33
33O040050	Gained	109	16	6.81	84	14	6
34C030030	Gained	136	21	6.48	119	19	6.26
34C030150	Gained	119	19	6.26	102	16	6.38
34C050030	Gained	139	20	6.95	91	15	6.07
34C100300	Lost	61	13	4.69	62	12	5.17
34D030800	Lost	99	16	6.19	108	20	5.4
34G010020	Gained	72	12	6	90	15	6
34G020200	Lost	95	16	5.94	62	11	5.64

34M020100	Gained	114	18	6.33	99	15	6.6
34O030200	Maintained	119	18	6.61	121	20	6.05
34S030050	Gained	90	16	5.63	97	16	6.06
34T010500	Gained	136	20	6.8	103	17	6.06
34Y010100	Maintained	124	19	6.53	102	16	6.38
34Y010400	Maintained	131	19	6.89	93	14	6.64
34Y020275	Lost	69	13	5.31	56	11	5.09
35C030200	Maintained	77	13	5.92	82	15	5.47
35D050020	Maintained	91	13	7	102	15	6.8
35D161000	Lost	123	19	6.47	72	12	6
35F010100	Lost	135	19	7.11	110	17	6.47
35G020200	Maintained	105	17	6.18	85	15	5.67
35G030100	Gained	121	18	6.72	104	16	6.5
35G040080	Maintained	107	16	6.69	92	14	6.57
36R020200	Maintained	104	17	6.12	100	18	5.56

Table D.5. List of BMWP, N-taxa and ASPT (BMWP) scores for the sixty-five sample sites during Spring 2017 and Summer 2017.

		Spring 2017			Summer 2017		
Station	**Status**	**BMWP**	**N-taxa**	**ASPT**	**BMWP**	**N-taxa**	**ASPT**
25A030100	Maintained	114	17	6.71	142	20	7.1
25B070200	Lost	126	19	6.63	103	17	6.06
25B100100	Gained	156	24	6.5	108	17	6.35
25B100200	Maintained	154	24	6.42	98	16	6.13
25B150050	Lost	121	20	6.05	94	17	5.53
25B150500	Maintained	97	16	6.06	103	19	5.42
25C030100	Gained	159	25	6.36	82	15	5.47
25C070200	Gained	136	23	5.91	116	20	5.8
25C090100	Lost	165	25	6.6	136	22	6.18
25C090400	Maintained	171	25	6.84	118	19	6.21
25D100200	Lost	126	19	6.63	102	17	6
25G040025	Maintained	138	21	6.57	121	20	6.05
25W010300	Lost	160	24	6.67	115	20	5.75
26D070700	Lost	87	16	5.44	97	17	5.71
26F020080	Gained	96	17	5.65	118	20	5.9
26F020250	Gained	94	16	5.88	147	23	6.39
26I030300	Maintained	87	15	5.8	96	16	6
26I030400	Lost	122	18	6.78	111	19	5.84
26K010300	Gained	159	26	6.12	199	33	6.03
26L030350	Gained	139	23	6.04	171	28	6.11
26S030400	Maintained	135	20	6.75			
26Y010200	Gained	97	15	6.47	124	20	6.2
27B020600	Lost	102	15	6.8	77	13	5.92
27G020600	Maintained	156	23	6.78	102	17	6
27O010700	Maintained	114	20	5.7			
29B020100	Lost	141	23	6.13	155	26	5.96
29B040300	Maintained	112	16	7	93	15	6.2
29O011000	Maintained	141	21	6.71	119	19	6.26
30G010250	Maintained	117	18	6.5	111	18	6.17
30N010100	Lost	132	21	6.29	142	23	6.17
31R010100	Lost	40	7	5.71	53	9	5.89
32B030050	Maintained	119	19	6.26	70	13	5.38
32C010020	Maintained	108	18	6	88	15	5.87
32C030150	Gained	115	17	6.76	128	21	6.1
32C050050	Maintained	103	17	6.06	124	21	5.9
32E010030	Gained	149	23	6.48	98	16	6.13
32G070300	Lost	107	17	6.29	96	15	6.4
32O040250	Gained	92	16	5.75	34	9	3.78
33A020100	Lost	57	10	5.7	107	19	5.63
33B010100	Lost	152	23	6.61	108	18	6
33G020200	Gained	111	18	6.17	100	16	6.25
33K010200	Maintained	153	23	6.65	113	18	6.28
33O040050	Gained	143	21	6.81	77	14	5.5
34C030030	Gained	194	29	6.69			
34C030150	Gained	166	25	6.64	176	29	6.07
34C050030	Gained	134	21	6.38	81	14	5.79
34C100300	Lost	96	17	5.65	104	18	5.78

34D030800	Lost	154	25	6.16	168	28	6
34G010020	Gained	136	21	6.48	88	14	6.29
34G020200	Lost	132	20	6.6	82	14	5.86
34M020100	Gained	113	18	6.28	97	16	6.06
34O030200	Maintained	159	24	6.63	165	26	6.35
34S030050	Gained	147	22	6.68	114	19	6
34T010500	Gained	159	23	6.91	132	20	6.6
34Y010100	Maintained	159	23	6.91	150	23	6.52
34Y010400	Maintained	150	22	6.82	116	19	6.11
34Y020275	Lost	125	20	6.25	115	20	5.75
35C030200	Maintained	92	16	5.75	102	18	5.67
35D050020	Maintained	125	20	6.25	99	17	5.82
35D161000	Lost	107	17	6.29	117	19	6.16
35F010100	Lost	152	25	6.08	96	15	6.4
35G020200	Maintained	132	22	6	141	22	6.41
35G030100	Gained	123	20	6.15	114	18	6.33
35G040080	Maintained	152	23	6.61	116	18	6.44
36R020200	Maintained	135	22	6.14	112	19	5.89

Table D.6. List of WHPT, N-taxa and ASPT (WHPT) scores for the sixty-five sample sites during Spring 2016 and Summer 2017.

		Spring 2016			Summer 2016		
Station	**Status**	**WHPT**	**N-taxa**	**ASPT**	**WHPT**	**N-taxa**	**ASPT**
25A030100	Maintained	197.4	25	7.9	112.8	16	7.05
25B070200	Lost	139.7	20	6.99	116.8	18	6.49
25B100100	Gained	205.3	29	7.08	145.8	21	6.94
25B100200	Maintained	202.3	28	7.23	150.9	23	6.56
25B150050	Lost	164.1	24	6.84	119.8	20	5.99
25B150500	Maintained	183.9	28	6.57			
25C030100	Gained	154.8	22	7.04	77.1	13	5.93
25C070200	Gained	213.2	28	7.61	116.5	18	6.47
25C090100	Lost	172.3	24	7.18	141.3	21	6.73
25C090400	Maintained	188.5	25	7.54	142.3	21	6.78
25D100200	Lost	147.6	22	6.71	121.7	18	6.76
25G040025	Maintained	184.3	26	7.09	180.2	26	6.93
25W010300	Lost	218.3	34	6.42			
26D070700	Lost	143.9	23	6.26	139.2	23	6.05
26F020080	Gained	130.9	21	6.23	128.1	20	6.41
26F020250	Gained	160.7	23	6.99	133.9	21	6.38
26I030300	Maintained	143.3	23	6.23	161	26	6.19
26I030400	Lost	130.7	21	6.22	148.2	21	7.06
26K010300	Gained	117	18	6.5	203	31	6.55
26L030350	Gained	173.5	26	6.67	217.9	32	6.81
26S030400	Maintained	135.2	23	5.88	117.3	23	5.1
26Y010200	Gained	135.7	20	6.79	74.6	13	5.74
27B020600	Lost	180.5	25	7.22	68.5	12	5.71
27G020600	Maintained	202.8	27	7.51	156	24	6.5
27O010700	Maintained	152.2	24	6.34	134.9	21	6.42
29B020100	Lost	190	30	6.33	171.6	29	5.92
29B040300	Maintained	181.8	26	6.99	162.2	23	7.05
29O011000	Maintained	167.5	22	7.61	157.2	23	6.83
30G010250	Maintained	91.5	15	6.1	117.6	19	6.19
30N010100	Lost	189.8	29	6.54	141.9	22	6.45
31R010100	Lost	100.2	14	7.16			
32B030050	Maintained	153.6	20	7.68	61.2	10	6.12
32C010020	Maintained	112.4	15	7.49	127.3	18	7.07
32C030150	Gained	184	27	6.81	155.3	23	6.75
32C050050	Maintained	182.1	25	7.28	130.2	20	6.51
32E010030	Gained	177	24	7.38	101	16	6.31
32G070300	Lost	143	20	7.15	132.9	21	6.33
32O040250	Gained	124.5	19	6.55	131.9	22	6
33A020100	Lost	130.7	20	6.54	120.3	19	6.33
33B010100	Lost	164.5	24	6.85	159.2	24	6.63
33G020200	Gained	134	19	7.05	161.6	24	6.73
33K010200	Maintained	206	28	7.36	124.4	18	6.91
33O040050	Gained	169.8	23	7.38	133.5	21	6.36
34C030030	Gained	198.4	28	7.09	194.8	27	7.21
34C030150	Gained	205.2	29	7.08	176.5	24	7.35
34C050030	Gained	208	28	7.43	144.6	21	6.89
34C100300	Lost	80.1	16	5.01	96.2	17	5.66
34G010020	Gained	110.7	17	6.51	108.3	17	6.37
34G020200	Lost	150.8	22	6.85	99.2	15	6.61
34M020100	Gained	181.3	26	6.97	163.7	23	7.12

34O030200	Maintained	171.4	25	6.86	182.3	28	6.51
34S030050	Gained	148.4	23	6.45	161.7	24	6.74
34T010500	Gained	193.5	26	7.44	174.4	26	6.71
34Y010100	Maintained	177.8	25	7.11	162.3	24	6.76
34Y010400	Maintained	203.6	28	7.27	144.2	21	6.87
34Y020275	Lost	113.6	20	5.68	99.5	17	5.85
35C030200	Maintained	124.6	18	6.92	131.5	21	6.26
35D050020	Maintained	140.4	19	7.39	153.6	21	7.31
35D161000	Lost	162.8	24	6.78	114.2	18	6.34
35F010100	Lost	171.8	23	7.47	145.1	21	6.91
35G020200	Maintained	154.6	23	6.72	135.1	20	6.76
35G030100	Gained	187.9	26	7.23	174.2	23	7.57
35G040080	Maintained	172.5	23	7.5	133.1	18	7.39
36R020200	Maintained	157	24	6.54	147.1	23	6.4

Table D.7. List of WHPT, N-taxa and ASPT (WHPT) scores for the sixty-five sample sites during Spring 2017 and Summer 2017.

		Spring 2017			Summer 2017		
Station	**Status**	**WHPT**	**N-taxa**	**ASPT**	**WHPT**	**N-taxa**	**ASPT**
25A030100	Maintained	160.8	21	7.66	161.9	21	7.71
25B070200	Lost	166.5	23	7.24	129.2	19	6.8
25B100100	Gained	199.3	28	7.12	119	17	7
25B100200	Maintained	198.5	27	7.35	123.8	19	6.52
25B150050	Lost	172	24	7.17	115.9	19	6.1
25B150500	Maintained	137.2	20	6.86	117.1	20	5.86
25C030100	Gained	208.9	29	7.2	94.1	16	5.88
25C070200	Gained	181.1	27	6.71	146.2	22	6.65
25C090100	Lost	203.4	28	7.26	154.4	23	6.71
25C090400	Maintained	210.1	29	7.24	139.1	20	6.96
25D100200	Lost	160.9	21	7.66	114	18	6.33
25G040025	Maintained	181.1	25	7.24	141.4	21	6.73
25W010300	Lost	190.5	27	7.06	147.6	25	5.9
26D070700	Lost	110.3	18	6.13	117.8	19	6.2
26F020080	Gained	128.5	21	6.12	134.3	21	6.4
26F020250	Gained	114	18	6.33	151.3	23	6.58
26I030300	Maintained	116.8	19	6.15	107.8	17	6.34
26I030400	Lost	145.1	20	7.26	138.2	23	6.01
26K010300	Gained	201.6	31	6.5	227.3	36	6.31
26L030350	Gained	172.8	26	6.65	202.3	31	6.53
26S030400	Maintained	143.8	21	6.85			
26Y010200	Gained	122.9	18	6.83	147.3	23	6.4
27B020600	Lost	148.7	19	7.83	81.9	13	6.3
27G020600	Maintained	199	27	7.37	120.1	18	6.67
27O010700	Maintained	140.1	22	6.37			
29B020100	Lost	163.1	26	6.27	173.3	29	5.98
29B040300	Maintained	155.8	21	7.42	117.3	17	6.9
29O011000	Maintained	181.3	25	7.25	147.9	21	7.04
30G010250	Maintained	159.9	23	6.95	122.7	18	6.82
30N010100	Lost	165.1	25	6.6	165.3	26	6.36
31R010100	Lost	55.1	8	6.89	64.8	10	6.48
32B030050	Maintained	153.8	22	6.99	72.3	13	5.56
32C010020	Maintained	143.3	21	6.82	107.7	16	6.73
32C030150	Gained	152.6	21	7.27	149	23	6.48
32C050050	Maintained	148.4	22	6.75	152	23	6.61
32E010030	Gained	188.3	26	7.24	114.1	17	6.71
32G070300	Lost	129	20	6.45	105.3	16	6.58
32O040250	Gained	109.4	18	6.08	55.9	11	5.08
33A020100	Lost	75.9	12	6.33	121.6	20	6.08
33B010100	Lost	199.2	27	7.38	121	19	6.37
33G020200	Gained	141.5	21	6.74	101.6	16	6.35
33K010200	Maintained	199.1	27	7.37	130.8	19	6.88
33O040050	Gained	179	25	7.16	96.5	16	6.03
34C030030	Gained	231.3	32	7.23			
34C050030	Gained	183.5	25	7.34	93.8	14	6.7
34C030150	Gained	218.6	30	7.29	217.4	33	6.59
34C100300	Lost	135.1	21	6.43	117	20	5.85
34D030800	Lost	206.5	30	6.88	198.8	30	6.63
34G010020	Gained	165.8	24	6.91	88.4	14	6.31
34G020200	Lost	156.3	22	7.1	96.4	15	6.43

34M020100	Gained	154.9	21	7.38	121.6	19	6.4
34O030200	Maintained	179.9	27	6.66	196	29	6.76
34S030050	Gained	189.9	26	7.3	152.2	23	6.62
34T010500	Gained	193.5	26	7.44	159.6	23	6.94
34Y010100	Maintained	204.7	27	7.58	152.2	23	6.62
34Y010400	Maintained	199.5	26	7.67	153.2	23	6.66
34Y020275	Lost	153.4	24	6.39	125.9	21	6
35C030200	Maintained	129.2	19	6.8	114.2	18	6.34
35D050020	Maintained	182.7	26	7.03	112.6	18	6.26
35D161000	Lost	152.8	22	6.95	128.5	20	6.43
35F010100	Lost	205.7	29	7.09	108.3	17	6.37
35G020200	Maintained	174.6	26	6.72	161.7	23	7.03
35G030100	Gained	161.6	23	7.03	130.1	19	6.85
35G040080	Maintained	200.7	27	7.43	140	20	7
36R020200	Maintained	173.7	26	6.68	146.7	22	6.67

Table D.8. List of Scope (%), Depth (cm) and Tile (score between 1-5) scores for the sixty-five sample sites during Spring 2016, Summer 2016, Spring 2017 and Summer 2017.

		Scope (%)				Depth (cm)				Tile (score bet. 1-5)			
Station_Code	**Status**	**Spring 2016**	**Summer 2016**	**Spring 2017**	**Summer 2017**	**Spring 2016**	**Summer 2016**	**Spring 2017**	**Summer 2017**	**Spring 2016**	**Summer 2016**	**Spring 2017**	**Summer 2017**
25A030100	Maintained	3.7	0.6	3	0.6	1.33	0.83	0.25	0	3	2.67	2.67	2.33
25B070200	Lost	0.5	1.6	0	0	0	0	0	0	1	1.67	1.33	1.67
25B100100	Gained	5.2	2.8	11	4.6	0.83	0.33	0	0	2.67	1.67	1.67	3
25B100200	Maintained	12	4.2	3	0.8	2.17	1.5	0	0	2.67	2.33	2.33	2.67
25B150050	Lost	18	45	6.6	2	1	0.17	0	0	3.67	3.33	3	3
25B150500	Maintained	20		24	13	1.17		0.17	0	2.33		3	3.33
25C030100	Gained	15	4.2	4.4	4.4	2	0.33	1.33	0	3	2.67	3.33	4.5
25C070200	Gained	22	3.4	8	1.6	0.33	0.67	0	0	2.67	3	2.33	3.33
25C090100	Lost		0.6	4	2.4		0.17	0	0		2	2	
25C090400	Maintained	0.2	25	18	1	0	0	0	0.67	2	2.33	3.67	
25D100200	Lost	11.2	20	4	4	0.67	0.17	0	0	3.33	1.67	3	2.33
25G040025	Maintained	0.2	0.4	0	0.2	0	0	0	0	2.33	3	2.33	3
25W010300	Lost	64		70	8	2.33		2.67	0.17	3.33		5	4.33
26D070700	Lost	21	5.4	24	9.4	1.33	0.83	6.83	0	1.67	4	4	3.67
26F020080	Gained	33	22	22	15	2.5	1	1.13		4.33	5	5	5
26F020250	Gained	52	27	53		0.75	1.5	0.88		3.33	4.67	4	
26I030300	Maintained	50	76	16	50	6.33	4.17	3.13		4.67	5	5	
26I030400	Lost	4.6	2.2	16	16		0.37	1	0.63	3	3.33	4.33	4
26K010300	Gained	1.6	2.6	21.8	11.6		0	0.65	0.67	3	3.33	3.33	4
26L030350	Gained	6.4	1.2	1	0		0	0	0	2	2.33	2	
26S030400	Maintained	6.25	60	41			1	1.25		5	5	5	
26Y010200	Gained	15.6		0.2	0.2	0.83		0	0	2.33	1.33	3.33	1.67
27B020600	Gained			0	0.4	2.5		0	0	3		2.33	2.33
27G020600	Gained	1.2	4.2	26	2.4	0.17	0	0.5	0	2.33	2.33	3	2
27O010700	Maintained	13		2		3.63		0		2.83		2.67	
29B020100	Lost	11.6	4.2	5	0.2	1	0	0.25	0	2	2.67	2.67	2

29B040300	Maintained	17.4	20.4	24		0.33	0.17	0.88	0	2.67	3	3	
29O011000	Maintained	27	4.2	31	44.8	5.3	0	1.38	4.5	2.4	3	2.67	3.67
30G010250	Maintained	5.2	4.8	1	0.6	0	0.17	0	0	1	1	1.67	1.33
30N010100	Lost	23.2	35	30	21.8	1.17	0.67	1.5	0.67	1.67	4.33	4.33	4
31R010100	Lost	0.2		0.2	0	0		0	0	1		0	1.67
32B030050	Maintained		1.8	4.8	1			0	0			2.33	
32C010020	Maintained	0.4	2.4	3.4	1.6	0	0	0	0	1	2.33	2.67	2.67
32C030150	Gained	2.3	0.6	5	1	3.1	0	0	0	1.5	2	2.67	2
32C050050	Maintained	23	15	20	15.6		0.17	0.38	0	1.67	3.33	4.33	3.33
32E010030	Gained	3.75	2.6	6	0	2.5	0	0		1		2.33	2
32G070300	Lost	4	6.6	4	4.2	0.5	0.17	3.38	0	2.33	2.67	3	2.67
32O040250	Gained	52	50	75	29	4.67	2	2.38	1.67	4.67	4.67	4.67	5
33A020100	Lost	2	3.6	1	0	0	0	0	0	1.33	2.33	2	2
33B010100	Lost		2	11	1		0.33	0	0	1.33	3.33	2	3
33G020200	Gained	12	10	16	0.4	0.67	0	0	0	2	2.67	3.67	3.67
33K010200	Maintained	3.6	1.4	1	0.8	0	0	0	0	1.33	1	1	2.33
33O040050	Gained	11	11	7	7	0.25	0	0	0	2	3	3	3.33
34C030030	Gained		1	2.4			0	0			3.33	3.67	
34C030150	Gained	3.67		0	0.4	1.5		0	0	2.5	1.67	3.33	1.67
34C050030	Lost		4.4	0	0		0	0	0		3	2.33	2
34C100300	Lost	54	30.2	24	65	5.83	2.17	1.63	5	5	5	3	5
34D030800	Lost	6.2	12	2	0.75	0	0.33	0	0	3	3.33	2.33	3.33
34G010020	Lost	4.5		5	1.2	0.83		1.63	0	1.33		1.33	2.33
34G020200	Gained	3	0	6	1	1.67	3.17	1.63	0	2.67	4.67	3	5
34M020100	Lost	2.4	1.8	3.6	4	0	0	0.13	0	2	2.67	3	2
34O030200	Maintained	14	16	4		1.17	1.5	0.63	0	4	4.67		
34S030050	Lost	26	8.6	28	22	0.67	0.5	0	0	4	3.67	4	3.67
34T010500	Gained	4.6	4	4	2	0	0	0.13	0	3.33	3	3.33	3.67
34Y010100	Maintained	30.6	0	3	1	1.67	0	0	0	2.67	2	2.33	2.67
34Y010400	Maintained	26	9	3.4	31.6	1	0.93	1.25	1.25	2	1.67	2	3
34Y020275	Gained	35	64	69	55	14.67	5.83	12	13.67	5	5	5	5
35C030200	Maintained	8	50	20	0.4	1	0.67	0	0	3.33	4.67	3.67	2

35D050020	Maintained	2.4	0.6	0	0.4	0	0	0	0	1.67	1.33	3.67	2.33
35D161000	Lost	4.4	21	28	15	0.67	0.17	1.5	1	3	2	3.33	4.67
35F010100	Lost	24	14	5.4		1.5	1.67	0	0	2.67	4	3.33	
35G020200	Maintained		9.2	2.8	0.2		0	0.5	0	2	3.67	2.33	2
35G030100	Gained			1.2				0				2	3.33
35G040080	Maintained	1.4	1.2	0	0.4	0.67	0	0	0	2.33	2.33	4.33	1.67
36R020200	Maintained	41	54	73	9	0	0.27	0.23	0.83	3	4	3	3

Table D.9. List of % Fine (%) and Quorer (g/m^2) scores for the sixty-five sample sites during Spring 2016, Summer 2016, Spring 2017 and Summer 2017.

		% Fine				Quorer (g/m^2)			
Station_Code	**Status**	**Spring 2016**	**Summer 2016**	**Spring 2017**	**Summer 2017**	**Spring 2016**	**Summer 2016**	**Spring 2017**	**Summer 2017**
25A030100	Maintained	5	12.5	7	8	0.83	0.31	0.34	0.08
25B070200	Lost	2	5	2	1	0.11	0.15	0.03	0.05
25B100100	Gained	1	5	3	5	0.22	0.07	0.05	0.16
25B100200	Maintained	2	20	6	2	0.5	0.13	0.12	0.09
25B150050	Lost	8	8	7	4	0.16	0.22	0.54	0.56
25B150500	Maintained	30		20	12	0.32		0.2	0.21
25C030100	Gained	7	0	7	10	0.22	0.14	0.89	0.32
25C070200	Gained	8	5	5	2	0.65	0.33	0.15	0.11
25C090100	Lost	0	0	7	2	0.05	0.16	0.05	0.07
25C090400	Maintained	2	10	10	2	0.2	0.18	0.11	0.1
25D100200	Lost	10	10	12	5	0.21	0.14	0.3	0.35
25G040025	Maintained	2	0	7	2	0.29	0.52	0.22	0.15
25W010300	Lost	35		40	18	0.97		0.67	0.19
26D070700	Lost	25	5	15	5	0.13	1.29	0.83	0.43
26F020080	Gained	30	15	25	5	1.9	1.47	1.64	0.13
26F020250	Gained	32.5	15	30	30	4.52	0.5	1.05	0.05
26I030300	Maintained	30	55	25	30	3.19	2.45	1.79	0.07
26I030400	Lost	17	10	8	17	0.46	0.23	0.51	0.36
26K010300	Gained	8	2	5	9	0.54	0.52	0.92	0.6
26L030350	Gained	5	2	5	2	0.03	0.04	0.19	0.04
26S030400	Maintained	27	10	23		0.94	0.86	0.61	
26Y010200	Gained	0	0	5	2.1	0.27	0.03	0.76	0.05
27B020600	Gained	5	0	9	2	1.13	0.13	0.06	0.23
27G020600	Gained	5	5	9	5	0.25	0.25	0.25	0.15
27O010700	Maintained	20		12		0.15		0.19	
29B020100	Lost	10	10	10	2	0.22	0.11	0.39	0.05

29B040300	Maintained	8	10	25	8	0.07	0.06	0.75	0.08
29O011000	Maintained	22	0	22	25	0.25	0.22	0.16	0.33
30G010250	Maintained	3	13	15	5	0.01	0.03	0.23	0.05
30N010100	Lost	25	30	35	25	0.09	0.95	0.79	0.39
31R010100	Lost	0		0	1	0.01		0.17	0.05
32B030050	Maintained	2	5	2	2	0.17	0.02	0.29	0.06
32C010020	Maintained	1	5	2	2	0.14	0.14	0.6	0.11
32C030150	Gained	3	3	3	2	0.05	0.07	0.15	0.06
32C050050	Maintained	10	15	8	8	0.16	0.28	0.46	0.49
32E010030	Gained	2	5	3	2	0.14	0.02	0.3	0.07
32G070300	Lost	2	5	4	8	0.17	0.25	0.33	0.13
32O040250	Gained	20	30	60	64	0.65	0.75	1.35	0.25
33A020100	Lost	5	5	5	2	0.22	0.19	0.2	0.21
33B010100	Lost	0	0	3	2	0.18	0.22	0.1	0.1
33G020200	Gained	5	5	10	2	0.17	0.21	0.39	0.11
33K010200	Maintained	0	0	0	2	0.06	0.01	0.22	0.01
33O040050	Gained	20	20	5	5	0.3	0.23	0.3	0.16
34C030030	Gained	0	2	7		0.36	0.14	0.08	
34C030150	Gained	3	3	6	6	0.14	0.24	0.43	0.11
34C050030	Lost	5	5	0	2	0.27	0.06	0.3	0.03
34C100300	Lost	50	50	20	55	4.28	0.85	0.74	0.18
34D030800	Lost	5	3	3	2	0.22	0.75	0.5	0.12
34G010020	Lost	30	30	20	30	0.22	0.03	0.23	0.05
34G020200	Gained	2	2	0	5	0.43	0.29	0.33	0.19
34M020100	Lost	5	5	5	2	0.14	0.1	0.24	0.03
34O030200	Maintained	15	0	10	5	0.45	0.48	0.4	0.11
34S030050	Lost	12	10	13	12	0.5	0.25	1.37	0.53
34T010500	Gained	5	10	7	2	0.24	0.13	0.2	0.1
34Y010100	Maintained	15	0	7	4	0.26	0.07	0.11	0.1
34Y010400	Maintained	20	25	5	15	0.09	0.13	0.33	1.33
34Y020275	Gained	60	25	60	50	6.3	0.48	3.1	0.21
35C030200	Maintained	5	25	15	2	0.63	0.98	0.74	0.16

35D050020	Maintained	3	0	2	2	0.26	0.16	0.29	0.28
35D161000	Lost	10	5	6	38	1.35	0.21	0.23	0.29
35F010100	Lost	10	20	6	6	1.34	0.62	0.37	0.14
35G020200	Maintained	0	0	2	2	0.6	0.14	0.14	0.09
35G030100	Gained	0	15	0	5	0.08	0.12	0.15	0.03
35G040080	Maintained	2	2	2	2	0.24	0.06	0.53	0.1
36R020200	Maintained	20	40	60	17	0.54	0.96	0.94	0.21

Appendix E

Table E.1. A list of invertebrates collected from the sixty-five sampling sites and the number of sites in which the taxa were present for each sampling period.

Phylum	Class	Order	Family	Taxon Name	Author	Num. of sites taxa present in Spring 2016	Sum 2016	Spring 2017	Sum 2017
Annelida	**Clitellaria - subclass Hirudinea**	**Arhynchobdellida**	Erpobdellidae	*Erpobdella octoculata*	(Linnaeus, 1758)	11	13	12	17
				Erpobdella testacea	(Savigny, 1812)	1			
			Hirudinidae	Haemopsis sanguiuga	(Linnaeus, 1758)		1		
		Rhynchobdellida	Glossiphoniidae	*Glossiphonia complanata*	(Linnaeus, 1758)	6	5	1	5
				Helobdella stagnalis	(Linnaeus, 1758)	6	6	1	7
			Piscicolidae	*Piscicola geometra*	(Linnaeus, 1761)		2	3	3
	Clitellaria - subclass Oligochaeta	**Olicochaeta**	Oligochaeta	Oligochaeta		62	60	65	59
Arthropoda	**Arachnida**	**Acari**	Beetle Mite	Oribatida		3			
			Mites	Hydrachnidae		32	33	38	21
	Entognatha	**Collembola**		*Furchula (springtail)*				1	1
	Insecta	**Coleoptera**	Chrysomelidae			1		2	
			Curculionidae			1	1	4	1
			Dytiscidae	Dytiscidae		1	2		
				Dytiscus	Linnaeus, 1758	1	1	5	4
				Oreodytes	Seidlitz, 1887		1		

	Oreodytes sanmarkii	(C.R. Sahlberg, 1826)	16	13	20	6
Elmidae	*Elmis aenea*	(Müller, 1806)	63	61	65	62
	Esolus parallelepipedus	(Müller, 1806)	59	59	63	55
	Limnius volckmari	(Panzer, 1793)	64	60	64	60
	Oulimnius sp.	Des Gozis, 1886	16	15		
	Oulimnius rivularis	(Rosenhauer, 1856)				1
	Oulimnius troglodytes	(Gyllenhal, 1827)	2			
	Oulimnius tuberculatus	(Müller, 1806)	30	22	16	18
Gyrinidae	Gyrinidae		1			
	Gyrinus	Müller, 1764		1		
	Orectochilus	Dejean, 1833	10	9	19	5
Haliplidae	*Brychius elevatus*	(Panzer, 1793)	2	4	5	1
	Haliplidae		1	2		1
	Haliplus	Latreille, 1802				1
Helophoridae	Helophoridae				4	
	Helophorus brevipalpis	Bedel, 1881	2			
Hydraenidae	Hydraenidae		2		3	
	Hydraena	Kugelann, 1794	2			
	Hydraena gracilis	Germar, 1824	38	44	38	34
	Limnebius truncatellus	(Thunberg, 1794)		1		
Hydrophilidae	Hydrophilidae			1	2	4
	Cercyon sp.	Leach, 1817	1			

	Scirtidae	*Elodes* sp.	Latreille, 1796	24	12	12	19
		Scirtes sp.	Illiger, 1807	7	4	10	6
Diptera	Athericidae	Athericidae		1	2		1
	Ceratopogonidae	Ceratopogonidae			1		
		Culicoides sp.	Latreille, 1809	13	4	16	17
		Dasyheleinae or Thaumaleidae		1		1	
	Chironomidae	Chironomidae		1	9		2
		Chironomini		21	41	45	34
		Chironomus					1
		Diamesinae		1	28	43	25
		Orthocladiinae			16	41	34
		Orthocladiinae / Diamesinae		61	23	1	
		Podonominae		36	48	45	42
		Prodiamesinae		1	35		4
		Tanypodinae		14	5	38	27
		Tanytarsini		44	57	57	40
	Culicidae	Culicidae			1		2
	Empididae	*Chelifera* sp.	Macquart, 1823	16	24	33	18
		Clinocerinae		10	8	5	1
		Empididae		21	14	32	13
		Hemerodromiinae		17	18	26	13
	Ephydridae	*Hydrellia* sp.	Robineau-Desvoidy, 1830				2
	Muscidae	*Limnophora* sp.	Robineau-Desvoidy, 1830	6	11	4	11
	Psychodidae	Pericomini		13	2	5	3

	Ptychopteridae	Ptychopteridae			2	1	2
	Simuliidae	*Prosimulium* sp.	Roubaud, 1906	6			
		Simuliidae		13	11	37	45
		Simulium sp.	Latreille, 1802	40	39	18	
	Stratiomyidae	*Stratiomyidae*				2	
	Limoniidae	*Antocha* sp.	Osten Sacken, 1860		3	1	
		Antocha vitripennis	(Meigen, 1830)	6	1	8	4
		Eloeophila sp.	Rondani, 1856	14	6	12	8
		Helius sp.	Le Peletier & Serville, 1828	1			1
		Limoniidae		2			
	Pediciidae	*Dicranota* sp.	Zetterstedt, [1838]	52	58	57	53
	Tipulidae	*Tipula* sp.	Linnaeus, 1758		5	3	5
Ephemeroptera	Baetidae	*Alainites muticus*	(Linnaeus, 1758)	58	57	61	41
		Baetidae		1	1		1
		Baetis sp.	Leach, 1815	1	1		
		Baetis rhodani	Pictet, 1845	66	61	63	57
		Baetis scambus	Eaton, 1870				5
	Caenidae	*Caenis rivulorum*	Eaton, 1884	44	20	51	8
	Ephemerellidae	*Serratella ignita*	(Poda, 1761)	29	53	51	48
	Ephemeridae	*Ephemera danica*	Müller, 1764	8	7	10	7
	Heptageniidae	Heptageniidae		2	19	7	2
		Ecdyonurus sp.	Eaton, 1868	40	50	41	41
		Electrogena sp.	(Curtis, 1834)			1	1
		Electrogena affinis	(Eaton, 1883)	1	1		
		Electrogena lateralis	(Curtis, 1834)	12	13	16	23

		Heptagenia sp.		2	1	1	
		Heptagenia sulphurea	(Müller, 1776)	23	25	31	17
		Rhithrogena sp.		57	21	61	23
		Rhithrogena semicolorata	(Curtis, 1834)	1			
	Leptophlebiidae	Leptophlebiidae		1			2
		Paraleptophlebia cincta	(Retzius, 1835)			9	1
Hemiptera	Aphelocheiridae	*Aphelocheirus aestivalis*	(Fabricius, 1794)	2	3	3	1
	Corixidae	Corixidae			1		
	Gerridae	Gerridae			1	1	
	Mesoveliidae	Mesoveliidae			3	4	
	Notonectidae	Notonecta			1		
	Veliidae	Veliidae			2		
Lepidoptera	Crambidae			1			
Megaloptera	Sialidae	*Sialis lutaria*	(Linnaeus, 1758)		2		1
Odonata - Zygoptera	Calopterygidae (= Agriidae)	*Calopteryx* sp.	Leach, 1815	1		1	1
	Chloroperlidae	*Chloroperla (= Siphonoperla) torrentium*	(Pictet, 1841)	45	6	47	10
Plecoptera		*Chloroperla tripunctata*	(Scopoli, 1763)	14	1	3	
	Leuctridae	*Leuctra* sp.	Stephens, 1836	9	1	8	
		Leuctra fusca	(Linnaeus, 1758)		58	14	55
		Leuctra inermis	Kempny, 1899	43	5	47	4
		Leuctra nigra	(Olivier, 1811)	3			1

	Nemouridae	*Amphinemura sulcicollis*	(Stephens, 1836)	39		27	
		Nemoura sp.	Latreille, 1796	2	3		9
		Nemoura avicularis	Morton, 1894		1		
		Protonemura sp.	Kempny, 1898	18	30	4	29
	Perlidae	*Dinocras cephalotes*	(Curtis, 1827)	3	7	7	5
		Perla bipunctata	Pictet, 1833	19	13	16	17
	Perlodidae	*Isoperla grammatica*	(Poda, 1761)	46	1	44	3
	Taeniopterygidae	*Brachyptera risi*	(Morton, 1896)	20		5	
Trichoptera	Beraeidae	*Beraea maurus*	(Curtis, 1834)			1	
		Beraea pullata	(Curtis, 1834)			2	
		Beraeidae			1		
	Glossosomatidae	*Agapetus* sp.	Curtis, 1834	34	25	41	30
		Agapetus fuscipes	Curtis, 1834	10			
		Glossosoma	Curtis, 1834	5	3	2	1
		Glossosoma boltoni	Curtis, 1834	9	12	8	6
		Glossosoma conformis	Neboiss, 1963	6	5	5	1
		Glossosomatidae		3	1	3	
	Goeridae	*Goera pilosa*	(Fabricius, 1775)	1			1
		Goeridae		3			
		Silo sp.	Curtis, 1830	10	24	9	7
		Silo nigricornis	(Pictet, 1834)	8	4	6	15
		Silo pallipes	(Fabricius, 1781)	31	19	25	20
	Hydropsychidae	*Hydropsyche* sp.	Pictet, 1834	10	30	3	7
		Hydropsyche angustipennis	(Curtis, 1834)	8	5	1	1

	Hydropsyche contubernalis	McLachlan, 1865		1		
	Hydropsyche instabilis	(Curtis, 1834)	16	2	17	5
	Hydropsyche pellucidula	(Curtis, 1834)	27	34	24	36
	Hydropsyche siltalai	Döhler, 1963	53	19	53	18
	Hydropsychidae		1	3		
Hydroptilidae	*Hydroptila* sp.	Dalman, 1819	12	14	30	4
	Ithytrichia sp.	Eaton, 1873	23	3	22	5
Lepidostomatidae	*Lasiocephala basalis*	(Kolenati, 1848)	1			
	Lepidostoma hirtum	(Fabricius, 1775)	29	20	29	16
Leptoceridae	*Athripsodes* sp.	Billberg, 1820	12	5	15	6
	Leptoceridae		3	2	1	
Limnephilidae	*Chaetopteryx villosa*	(Fabricius 1798)	7		7	6
	Drusus annulatus	(Stephens, 1837)	3	3	10	3
	Halesus sp.	Stephens, 1836	1			1
	Halesus digitatus	(von Paula Schrank, 1781)	3			
	Halesus radiatus	(Curtis, 1834)	17	6	12	5
	Limnephilidae		17	17	2	11
	Limnephilus sp.	Leach in Brewster, 1815			2	
	Limnephilus lunatus	Curtis, 1834	3		1	
	Potamophylax sp.	Wallengren, 1891	1	1	1	1
	Potamophylax cingulatus	(Stephens, 1837)	5	1	2	
	Potamophylax latipennis	(Curtis, 1834)	8	2	3	

	Ecclisopteryx guttulata	(Pictet, 1834)		1		1
Odontoceridae	*Odontocerum albicorne*	(Scopoli, 1763)	9	9	8	11
Philopotamidae	*Chimarra marginata*	(Linnaeus, 1761)	3	3	3	4
	Philopotamus montanus	(Donovan, 1813)	3	4	2	4
	Wormaldia sp.	McLachlan, 1865	2	3	2	
Polycentropodidae	*Holocentropus dubius*	(Rambur 1842)	1			
	Plectrocnemia sp.	Stephens, 1836		1		
	Plectrocnemia conspersa	(Curtis, 1834)	3	4	25	9
	Plectrocnemia geniculata	McLachlan, 1871	19	1	5	2
	Polycentropodidae		2			
	Polycentropus sp.	Curtis, 1835	1	2		
	Polycentropus flavomaculatus	(Pictet, 1834)	9	29	8	25
	Polycentropus kingi	McLachlan, 1881	8	9	2	1
Psychomyiidae	*Lype phaeopa*	(Stephens 1836)		1		
	Lype reducta	(Hagen 1868)	5	1	3	1
	Metalype fragilis	(Pictet, 1834)	1		3	
	Psychomyia pusilla	(Fabricius, 1781)	8	2	20	1
	Psychomyiidae			1		1
	Tinodes maculicornis	(Pictet 1834)	1			
	Tinodes waeneri	(Linnaeus, 1758)				1
Rhyacophilidae	*Rhyacophila* sp.	Pictet, 1834	6	3	2	6
	Rhyacophila dorsalis	(Curtis, 1834)	34	46	36	39

				Rhyacophila munda	(McLachlan, 1862)	17	8	17	5
			Sericostomatidae	*Sericostoma personatum*	(Spence in Kirby and Spence, 1826)	31	27	41	30
	Malacostraca (subclass)	**Amphipoda**	Crangonyctidae	*Crangonyx pseudogracilis*	Bousfield, 1958	2	1	1	2
			Gammaridae	*Gammarus* sp.	Fabricius, 1775		1		
				Gammarus duebeni	Liljeborg, 1852	63	62	62	58
		Decapoda	Astacidae	*Austropotamobius pallipes*	(Lereboullet, 1858)	1	2		1
		Isopoda	Asellidae	*Asellus aquaticus*	(Linnaeus, 1758)	15	13	14	16
				Asellus meridianus	Racovitza, 1919	1	3		
Mollusca	**Bivalvia**	**Veneroida**	Sphaeriidae (Pea mussels)	*Pisidium* sp.	C. Pfeiffer, 1821	25	31	20	25
	Gastropoda	**Gastropoda**	Ancylidae	*Ancylus fluviatilis*	O.F. Müller, 1774	8	18	23	25
				Bithynia	Leach, 1818				1
			Bithyniidae	*Bithynia leachi*	(Sheppard, 1823)	3	1	2	
				Bithynia tentaculata	(Linnaeus, 1758)		2		
				Bithyniidae		3	3	2	1
			Hydrobiidae	*Potamopyrgus antipodarum*	(J.E.Gray, 1843)	31	40	32	42
				Lymnaea sp.	Lamarck, 1799		1		
				Lymnaea peregra (Radix peregra)	(Müller, 1774)	2	7	2	9
			Lymnaeidae	*Lymnaea stagnalis*	(Linnaeus, 1758)		1		
			Neritidae	*Theodoxus fluviatilis*	(Linnaeus, 1758)	3	4	5	3
			Physidae	Physidae					3

			Planorbidae	*Bathyomphalus contortus*	(Linnaeus, 1758)	1	2	1	4
				Planorbidae			3		
				Planorbis carinatus	O.F. Müller, 1774		3		1
				Planorbis planorbis	(Linnaeus, 1758)		1		2
			Valvatidae	*Valvata* sp.	O. F. Müller, 1774			2	
				Valvata cristata	O.F. Müller, 1774				1
				Valvata macrostoma	Morch, 1864	2	1		
				Valvata piscinalis	(O.F. Müller, 1774)		1		2
Nematoda		**Nematoda**	Nematoda	Nematoda		1			
		Nematoda		Nematode					1
Nematomorpha		**Nematomorpha (Horse hair worm)**		Nematomorpha (Horse hair worm)			5		4
		Tricladida	Dugesiidae	*Dugesia*	Girard, 1851	1	3	4	2
Platyhelmintes	**Turbellaria**			*Dugesia polychroa*	(Schmidt, 1861)			1	
			Planariidae	*Planaria torva*	(Müller OF, 1773)	1			
				Planariidae			1	1	
				Polycelis sp.	Ehrenberg, 1831		2	1	
				Polycelis felina	(Dalyell, 1814)	4	4	6	3
				Polycelis nigra	(Müller OF, 1773)	2	1		

Appendix F

Photographs of invertebrates collected from sampling sites

Plate 1 – Ephemeroptera (Mayfly)

a) *Baetis rhodani* Pictet, 1845

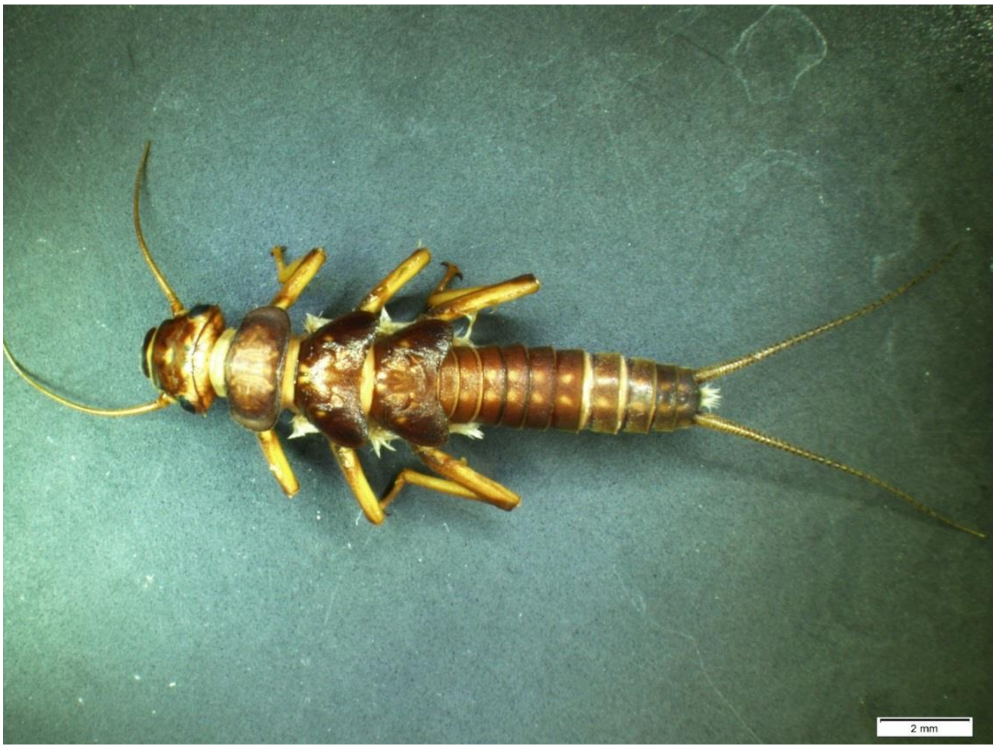
b) *Ecdyonorus* sp.

c) *Ephemera danica* Müller, 1764

d) *Rhithrogena semicolorata* (Curtis, 1834)

e) *Serratella ignita* (Poda, 1761)

Plate 2 – Plecoptera (Stonefly)



a) Diaocras cephalotes (Curtis, 1827)

b) *Perla bipunctata* Pictet, 1833

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Plate 3 – Plecoptera (Stonefly)

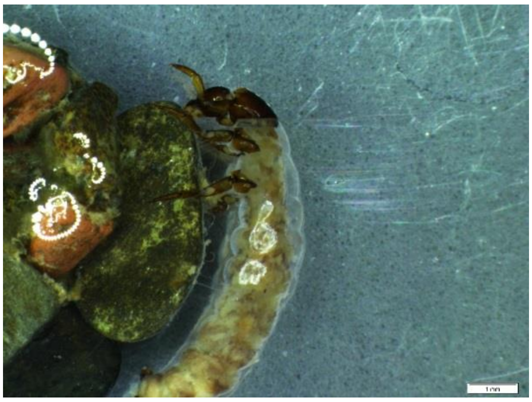
a) *Brachyptera risi* (Morton, 1896)

b) *Siphonoperla (=Chloroperla) torrentium* (Pictet, 1841)

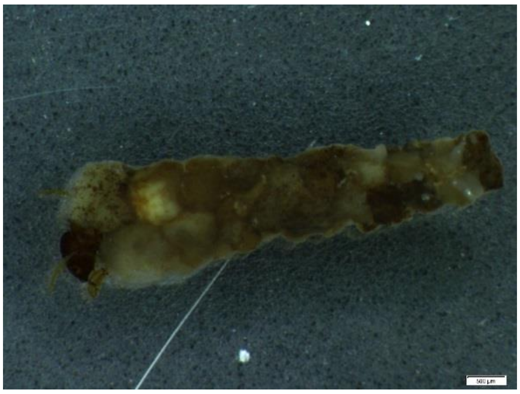
c) *Isoperla grammatica* (Poda, 1761)

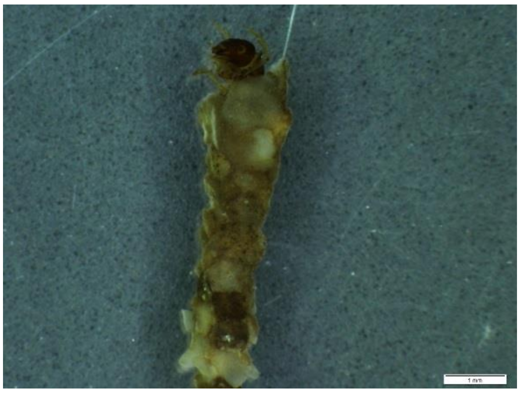
Plate 4 – Trichoptera (Caddisfly)

a) *Beraea maurus* (Curtis, 1834) b) *Chimarra marginata* (Linnaeus, 1761)



c) Glossosomatidae d) *Hydropsyche siltalai* Döhler, 1963





d) *Lepidostoma hirtum* (Fabricius, 1775) f) Lepidostoma hirtum (Fabricius, 1775)

Plate 5 – Trichoptera (Caddisfly)

a) Rhyacophila dorsalis (Curtis, 1834)

b) Sericostoma personatum (Spence in Kirby and Spence, 1826)

Plate 6 – Coleoptera (Beetles)

a) *Elmis aenea* (Müller, 1806)

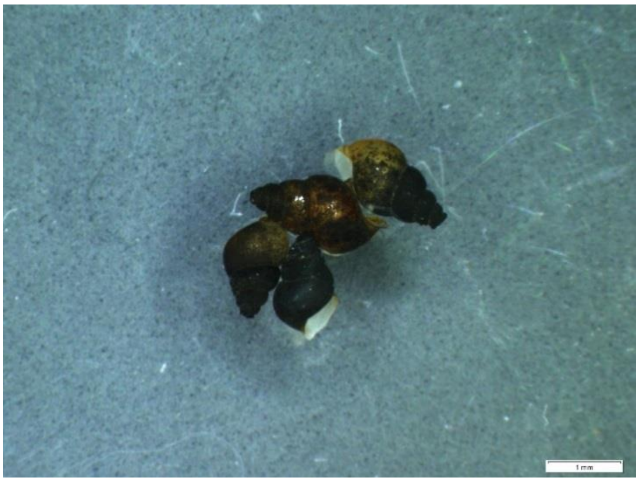
b) *Limnius volckmari (Panzer, 1793)*

c) Brychius elevatus (Panzer, 1793)

d) Gyrinidae

Plate 7 – Leech and Mollusca

a) *Haemopsis sanguiuga* (Linnaeus, 1758)



b) *Potamopyrgus antipodarum* (J.E.Gray, 1843)

c) *Theodoxus fluviatilis* (Linnaeus, 1758)

Appendix G

A List of dichotomous keys used in the identification of macro-invertebrates

General – Acari, Coleoptera (larvae), Diptera, Gammaridae, Hirudinea, and Oligochaeta

Fitter, R. and Manuel, R. (1986). *Collins field guide to Freshwater life of Britain and North-West Europe*. Collins, London.

Dobson, M., Pawley, S., Fletcher, M. & Powell, A. (2012). *Guide to Freshwater Invertebrates*. Freshwater Biological Association Scientific Publication No. 68, Ambleside, U.K. 216 pp.

Invertebrate Ireland Online. (2006). Checklist options. http://www.habitas.org.uk/invertebrateireland/checklistoptions.html

Nilsson, W. (2005). Nilsson, A. (ed.): Aquatic insects of Northern Europe. A taxonomic handbook, Vol. 1. 440 pp., hardbound, Apollo Books, DK – 5771 Stenstrup, Kirkeby Sand 19.

Nilsson, W. (2005). Nilsson, A. (ed.): Aquatic insects of Northern Europe. A taxonomic handbook, Vol. 2. 440 pp., hardbound, Apollo Books, DK – 5771 Stenstrup, Kirkeby Sand 19.

Coleoptera

Friday, L.E. (1988). *A key to the adults of British water beetles*. Field Studies, 7, 1–151.Available at: https://fsj.field-studies-council.org/media/797162/water_beetles_web.pdf

Holland, D. G. (1972). A *key to the larvae, pupa and adults of the British species of Elminthidae*. Freshwater Biological Association (FBA), scientific publication no. 26.

Ephemeroptera

Elliott, J. M. and Humpesch, U. H. (2012). *Mayfly larvae (Ephemeroptera) of Britain and Ireland: keys and a review of their ecology*. Freshwater Biological Association (FBA), scientific publication no. 66, Cumbria.

Hirudinea (Leeches)

Elliott, J. M. and Mann, K. H. (1979). *A key to the British freshwater leeches with notes on their life cycle and ecology*. Freshwater Biological Association (FBA), scientific publication no. 40, Cumbria.

Plecoptera

Elliott, J. M. (No year). Key to the larvae of British Stoneflies (Plecoptera). Freshwater Biological Association (FBA). http://www.fba.org.uk/recorders/publications_resources/keys/stonefly/contentParagraph/01/document/KeyToNymphsOfStoneflies.pdf Accessed online May 2011.

Feeley, H.B., Baars, J. R. & Kelly-Quinn, M. (2016) The Stonefly (Plecoptera) of Ireland – Distribution, Life Histories & Ecology. National Biodiversity Data Centre, Waterford. Ireland.

Hynes, H. B. N. 1977. *A key to the adults and nymphs of the British Stoneflies (Plecoptera) with notes on their ecology and distribution*. 3rd edition. Freshwater Biological Association (FBA), scientific publication no. 17.

Pryce, D., Macadam, C. and Brooks, S. 2007. *Guide to the British Stonefly (Plecoptera) families: adults and larvae*. Field Studies Council (FSC), Shropshire.

Trichoptera

Edington, J. M. and Hildrew, A. G. 1981. *Caseless Caddis larvae of the British Isles with notes on their ecology*. Freshwater Biological Association (FBA), scientific publication no. 43, London.

Wallace, I. D., Wallace, B. and Philipson, G. N. 2003. *Keys to the case-bearing caddis larvae of Britain and Ireland*. Freshwater Biological Association (FBA), scientific publication no. 61, Liverpool.

Wallace, I. 2006. *Simple key to Caddis larvae*. 1st edition, FSC, Liverpool.

Triclads (Flatworms)

Reynoldson, T. B. (1978). *A key to British species of freshwater Triclads.* Freshwater Biological Association (FBA), scientific publication no. 23, Cumbria.

Appendix H

1. Route determination/planner

To assess the best route between a starting location, e.g. Tuam, Co. Galway, and nearby sampling sites, a road network grid was created using Open Street Map layers for Ireland and Northern Ireland. This Open Street Map road layer was cleaned so as to remove any routes that were not drivable, i.e. cycle-ways, disused tracks, paths, pedestrian zones, footways, etc. Length and time fields were then generated for the newly modified road layer, so as to enable total time and costings for the journey.

In ArcMap GIS, clusters of six or seven closely located sample sites were selected, and using Tuam as the starting and end point (with the exception of Route 1 where Portstewart was used as the starting point), the Network Analyst tool-set was employed to determine the best route between sample sites. As there was sixty-five sample sites to visit, this resulted in the generation of thirteen route maps. These thirteen routes are presented Figures H.1-H.13.

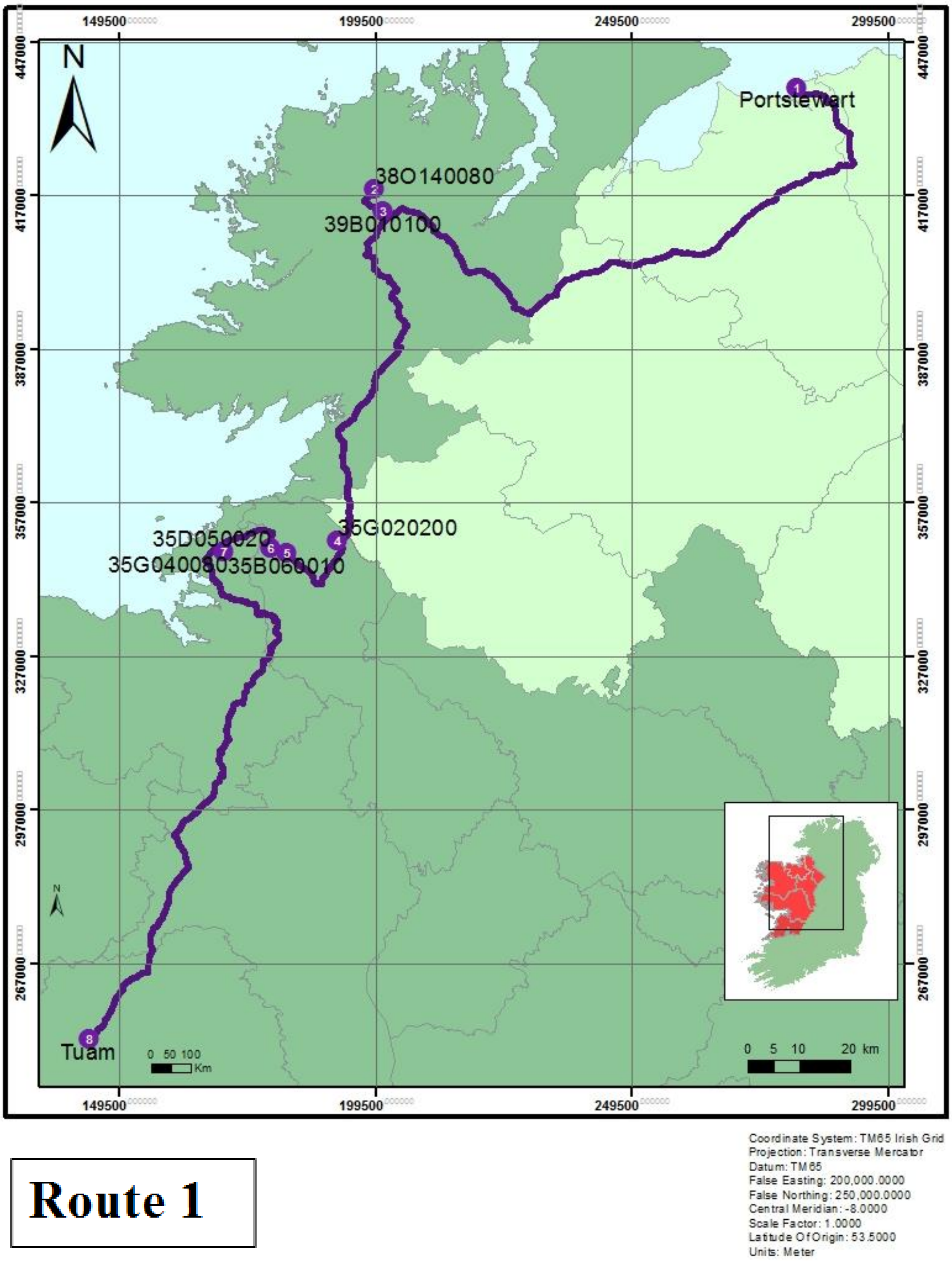


Figure H.1. Sample sites route map (Route 1).

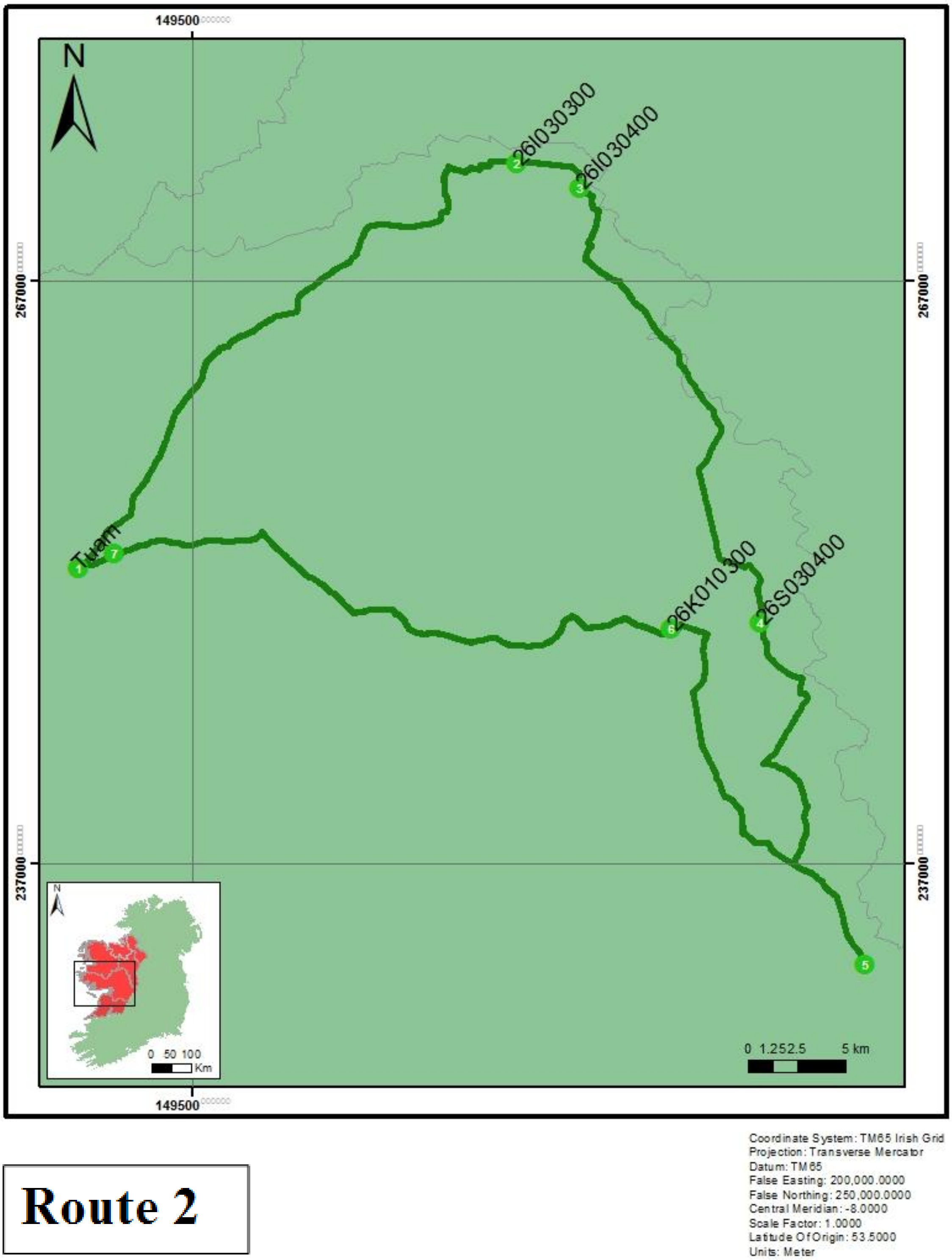


Figure H.2. Sample sites route map (Route 2).

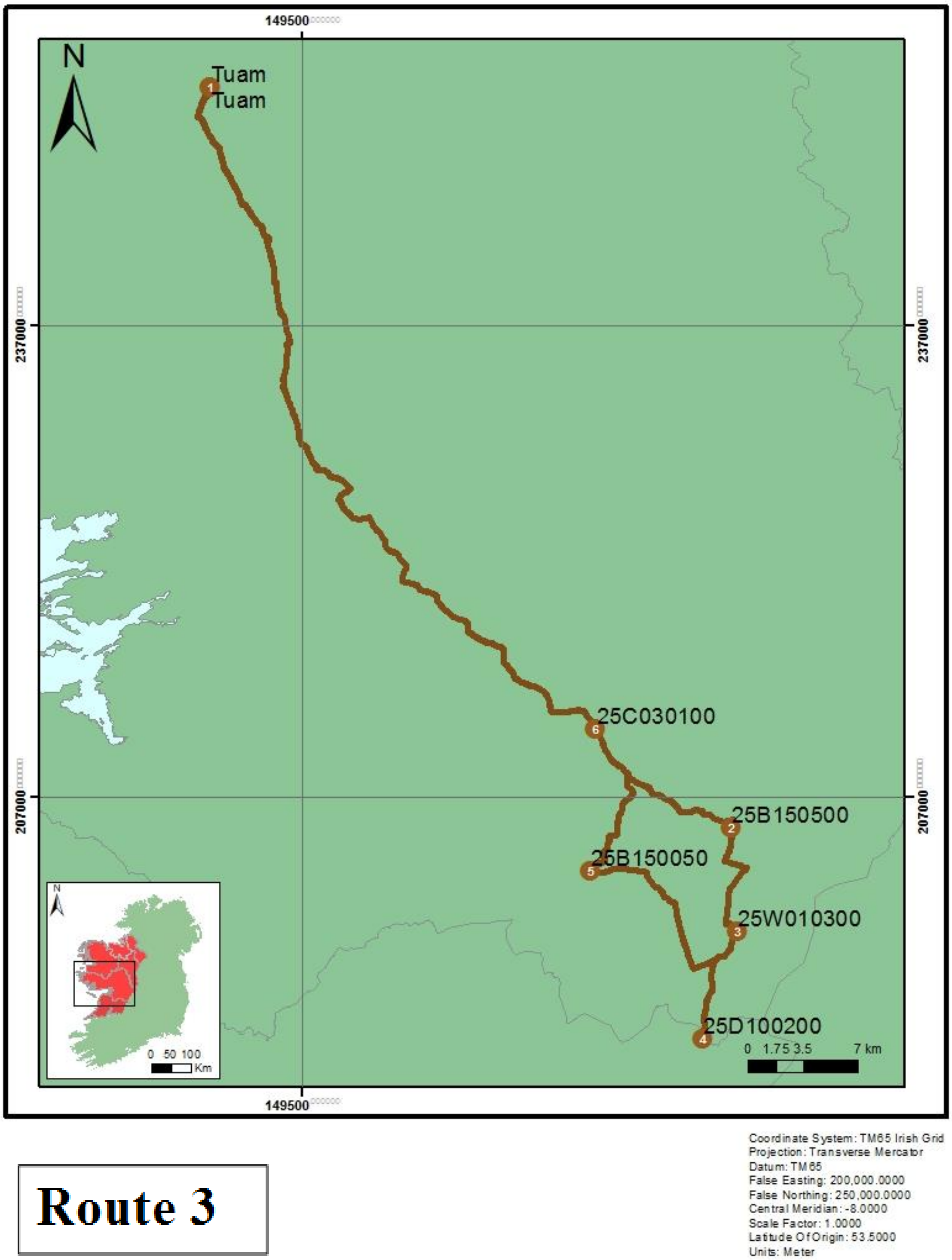


Figure H.3. Sample sites route map (Route 3).

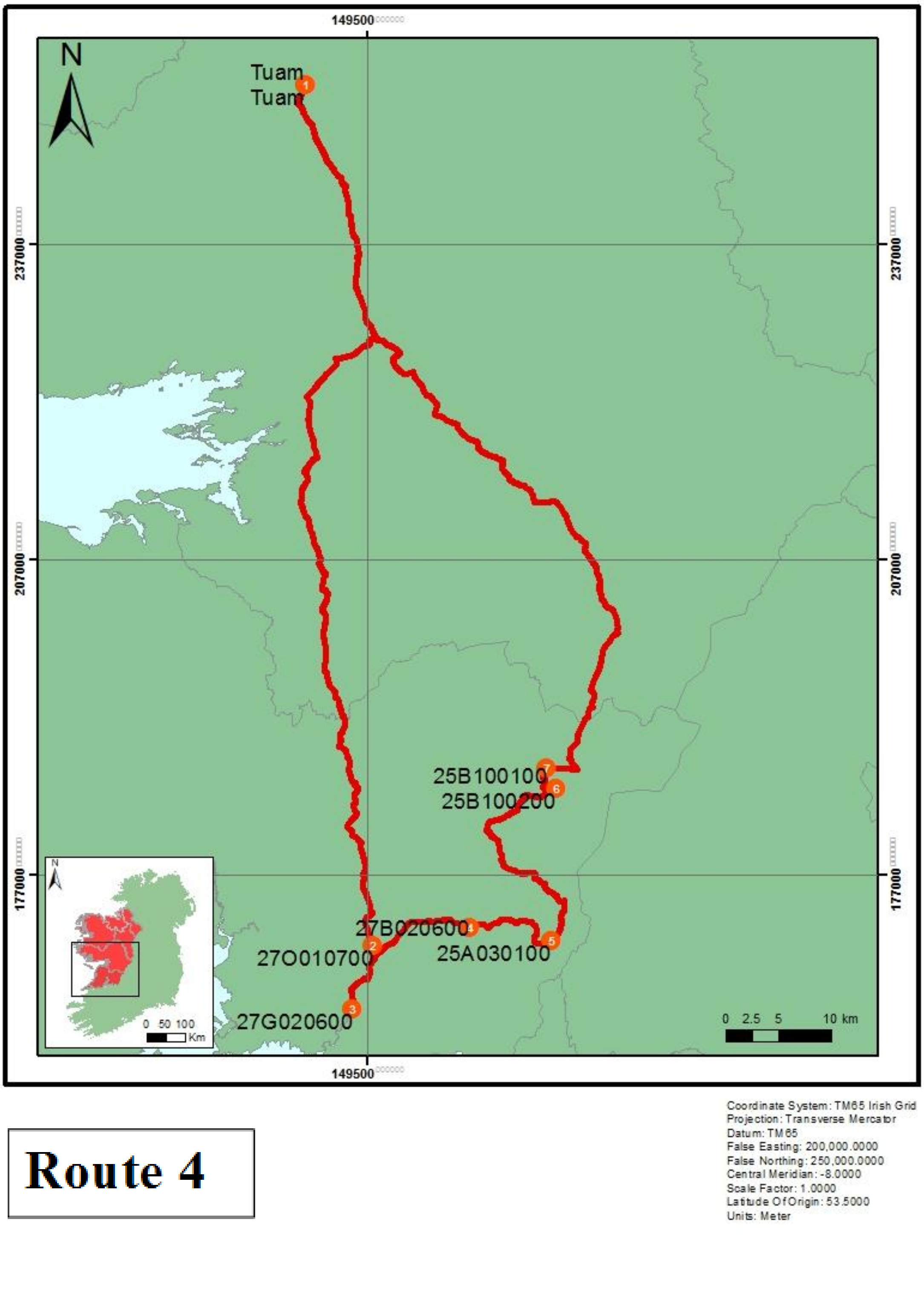


Figure H.4. Sample sites route map (Route 4).

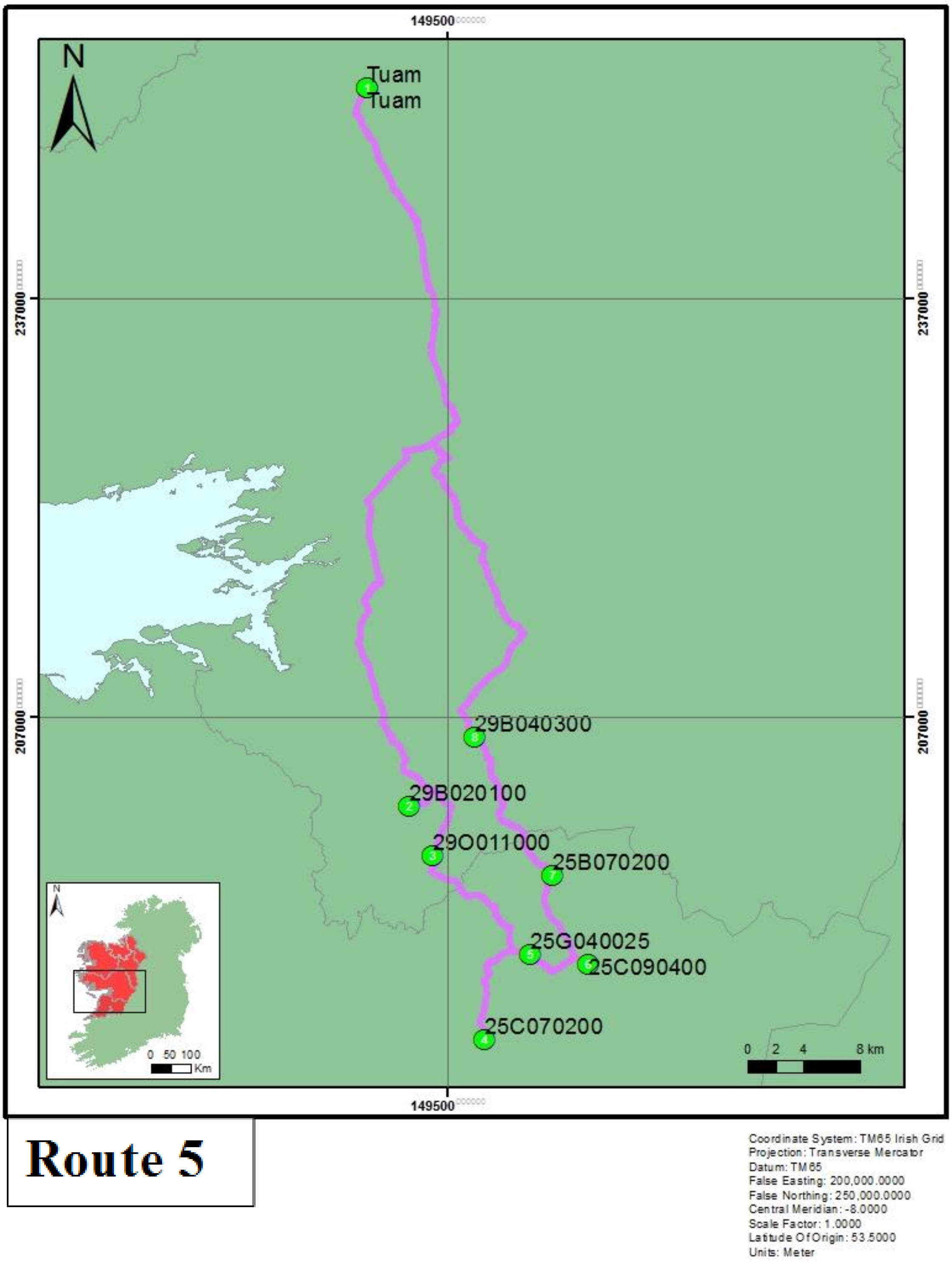


Figure H.5. Sample sites route map (Route 5).

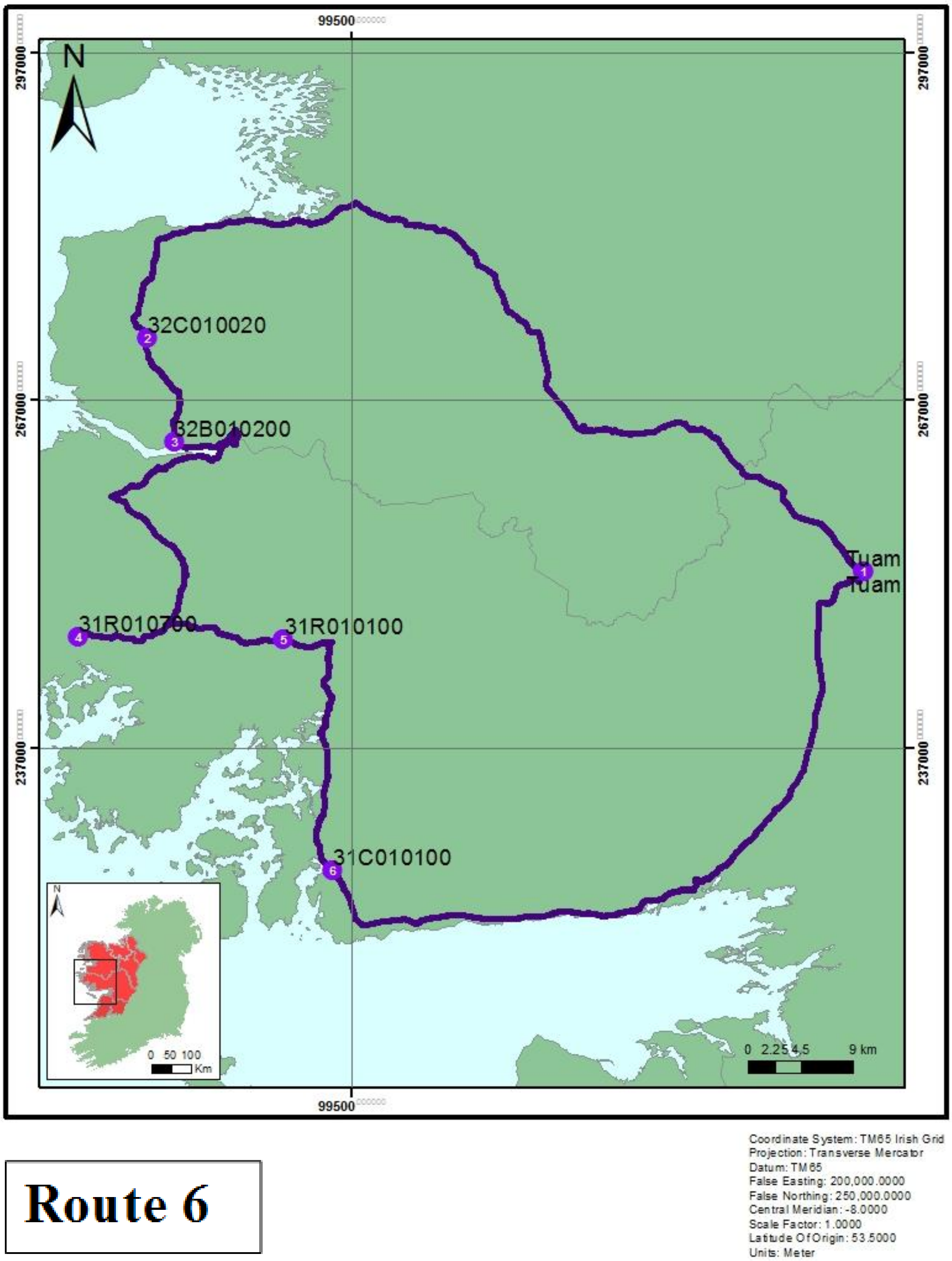


Figure H.6. Sample sites route map (Route 6).

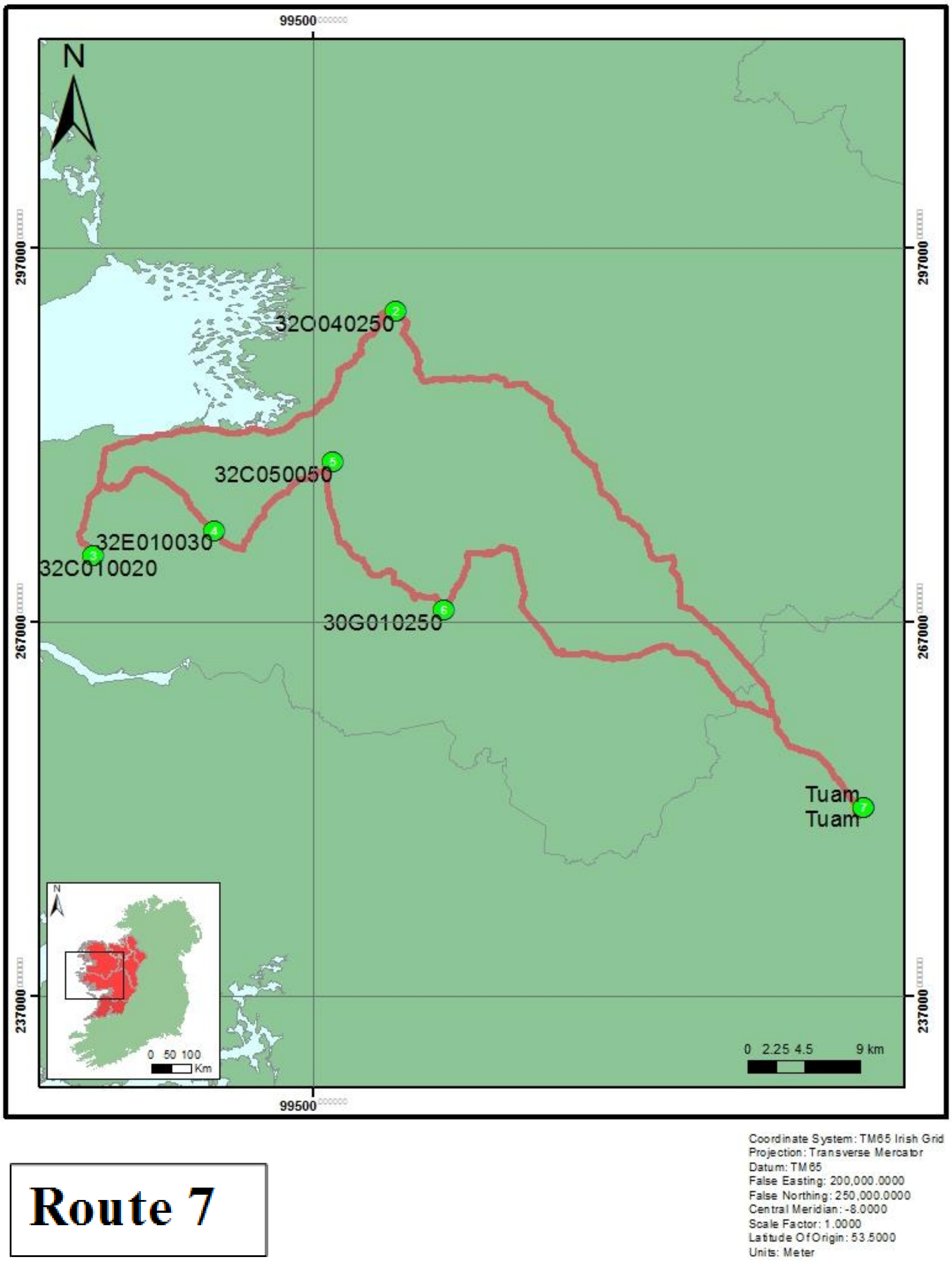


Figure H.7. Sample sites route map (Route 7).

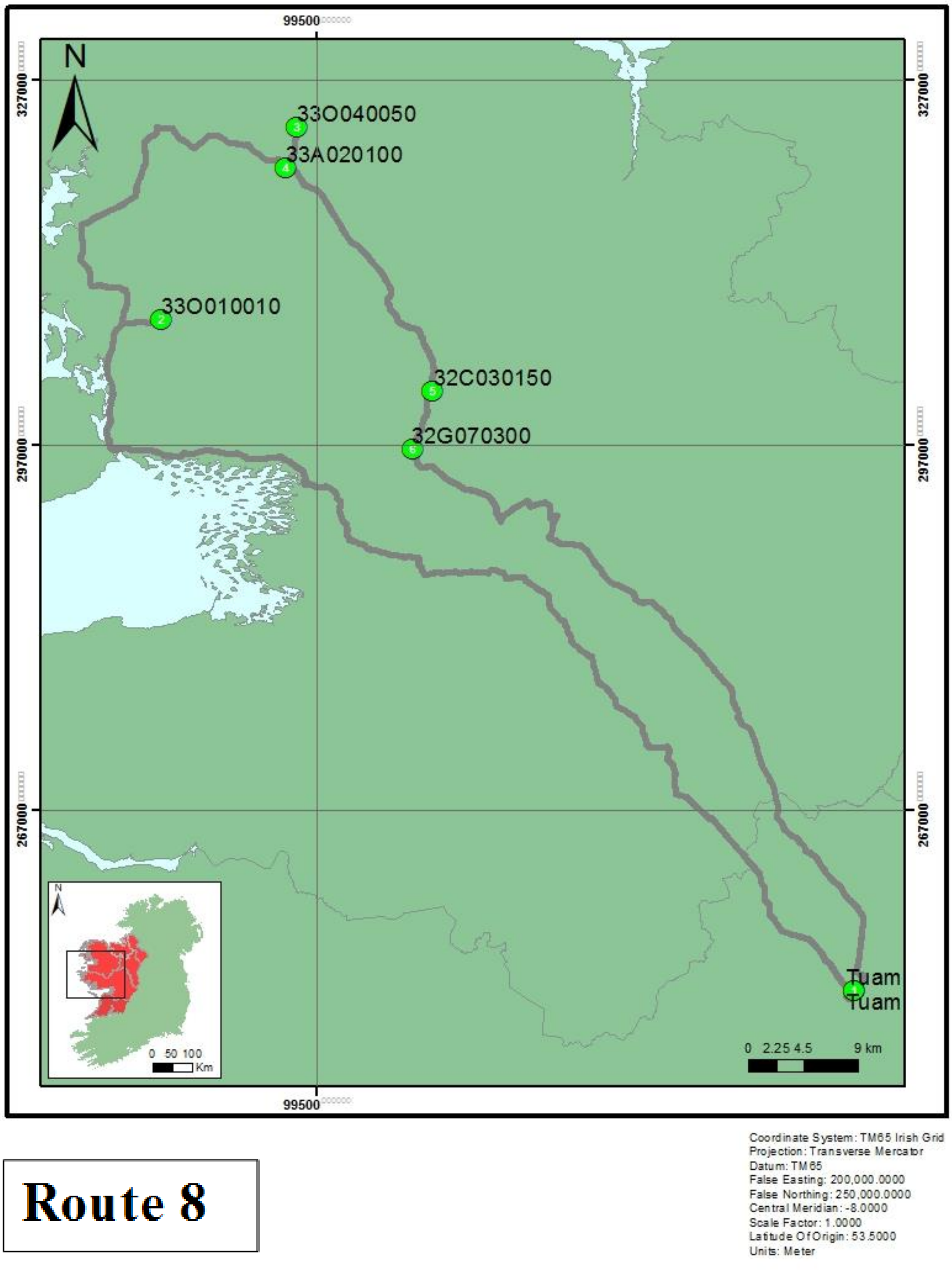


Figure H.8. Sample sites route map (Route 8).

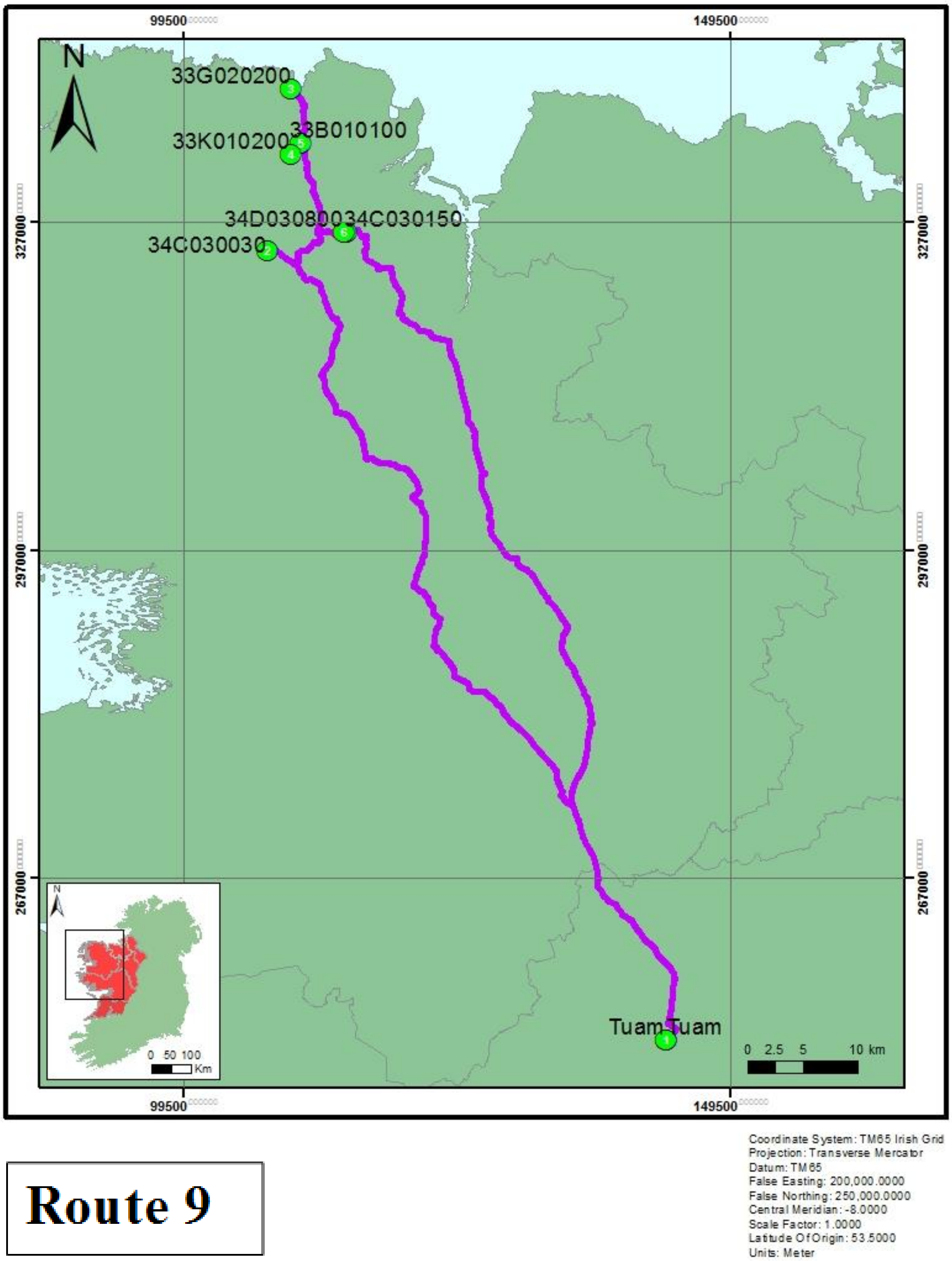


Figure H.9. Sample sites route map (Route 9).

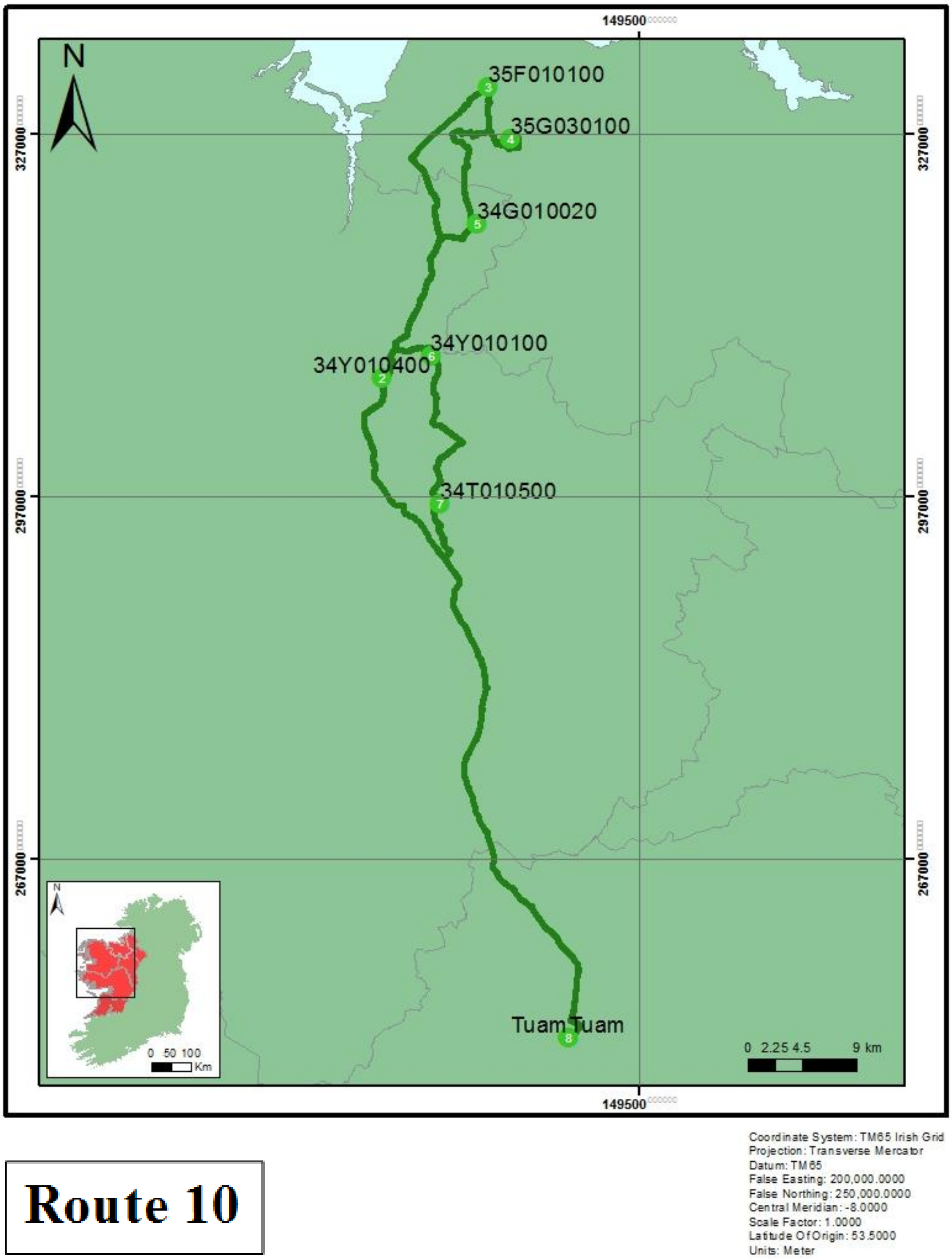
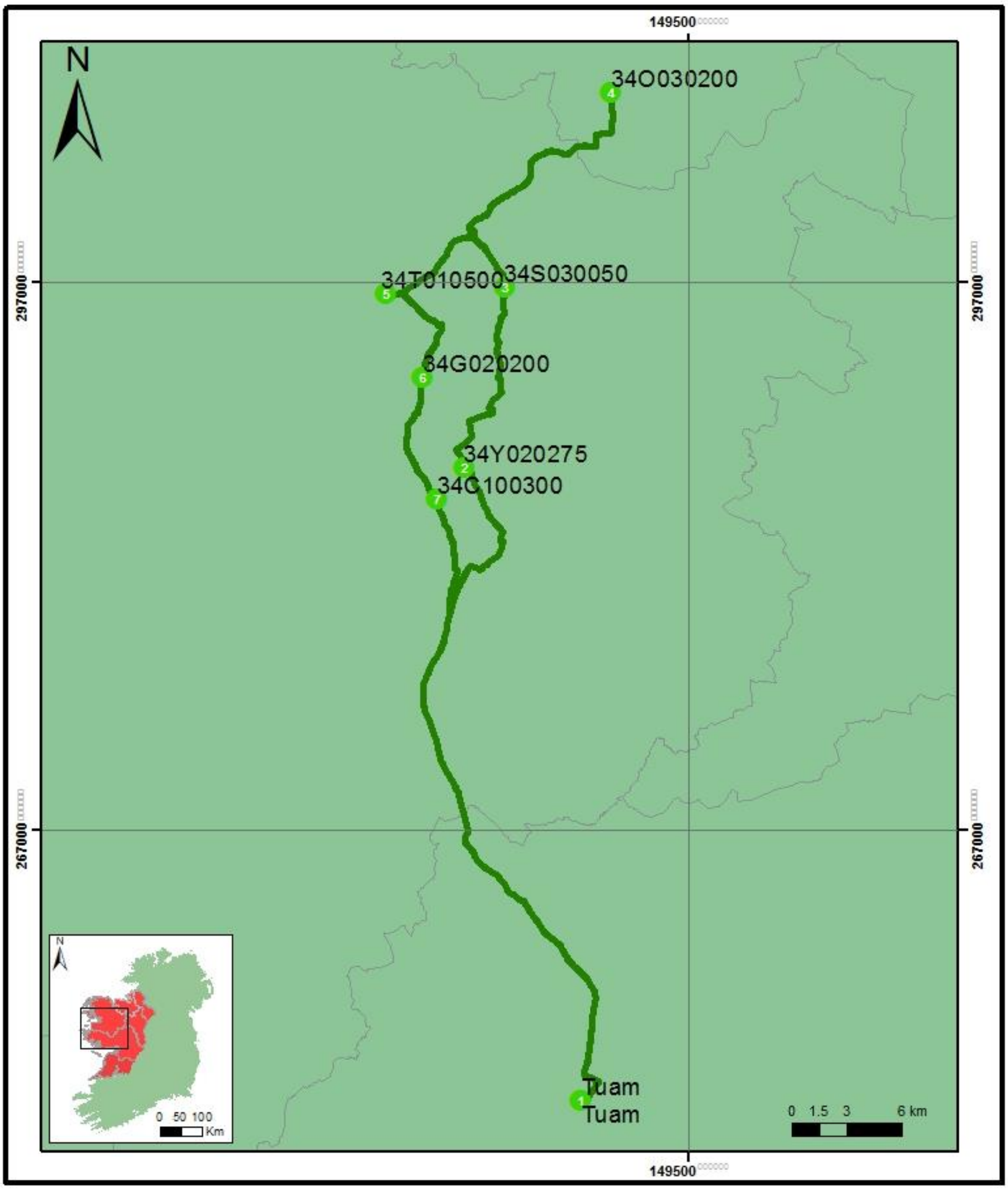


Figure H.10. Sample sites route map (Route 10).



Route 11

Coordinate System: TM65 Irish Grid
Projection: Transverse Mercator
Datum: TM65
False Easting: 200,000.0000
False Northing: 250,000.0000
Central Meridian: -8.0000
Scale Factor: 1.0000
Latitude Of Origin: 53.5000
Units: Meter

Figure H.11. Sample sites route map (Route 11).

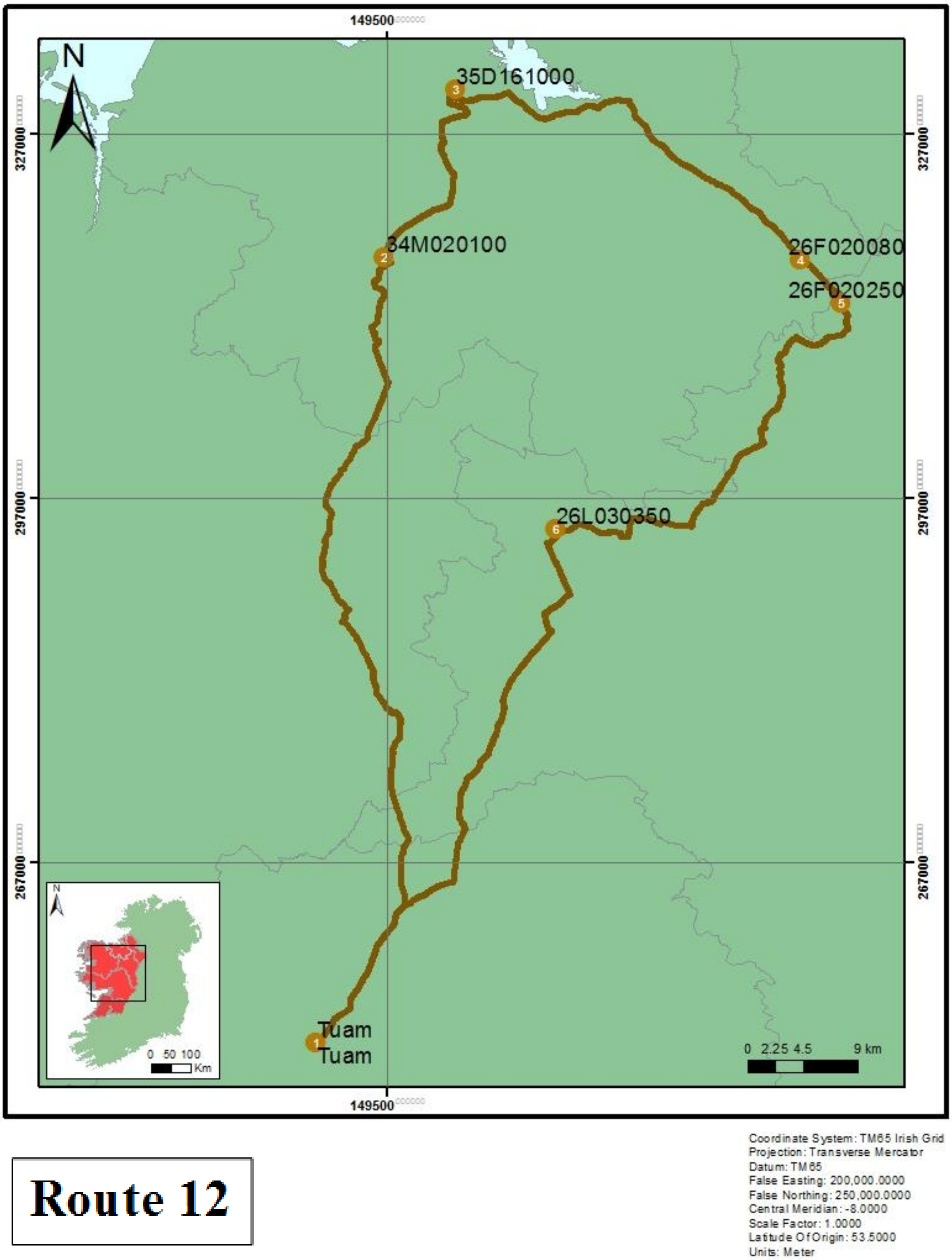


Figure H.12. Sample sites route map (Route 12)

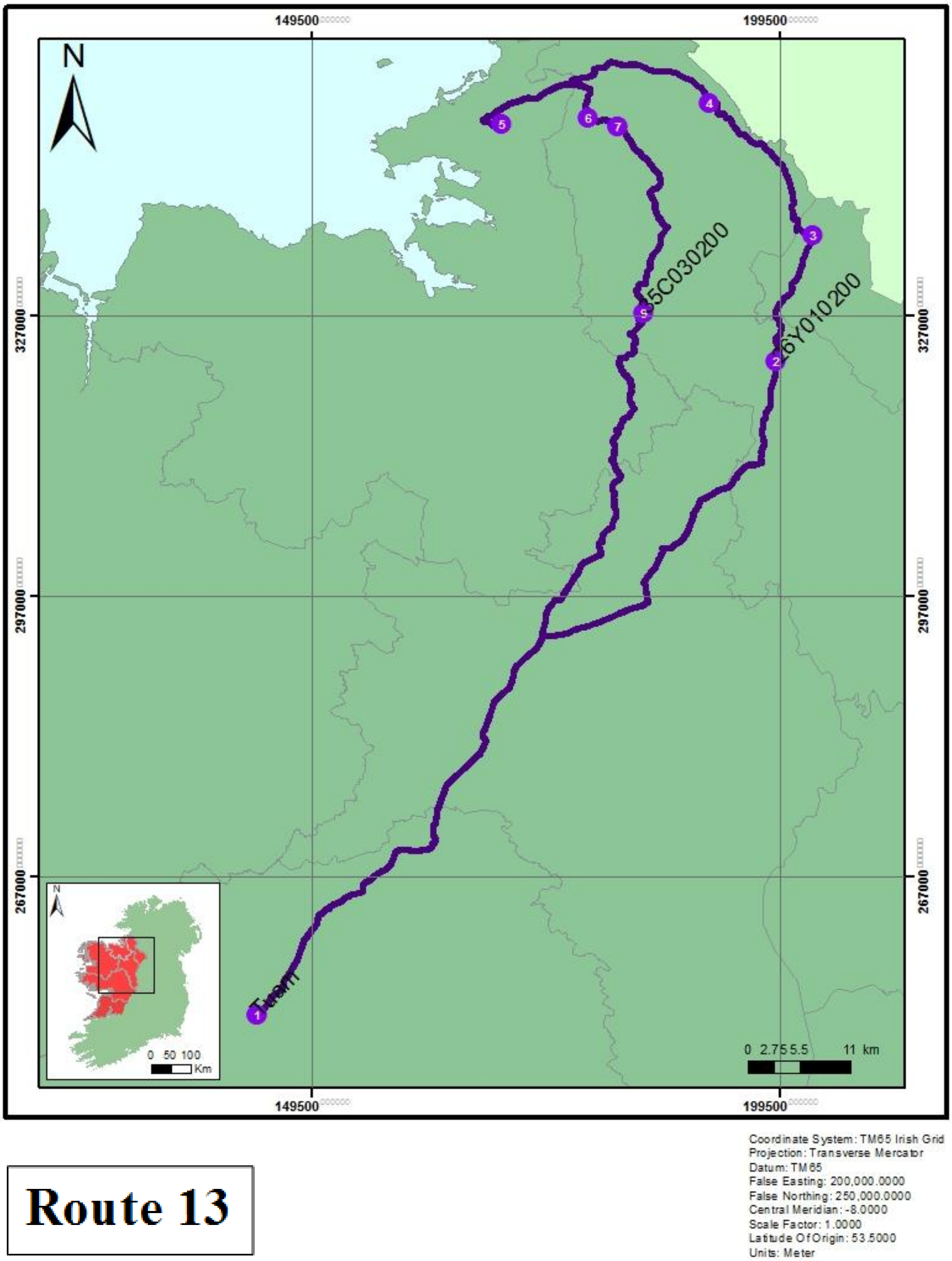
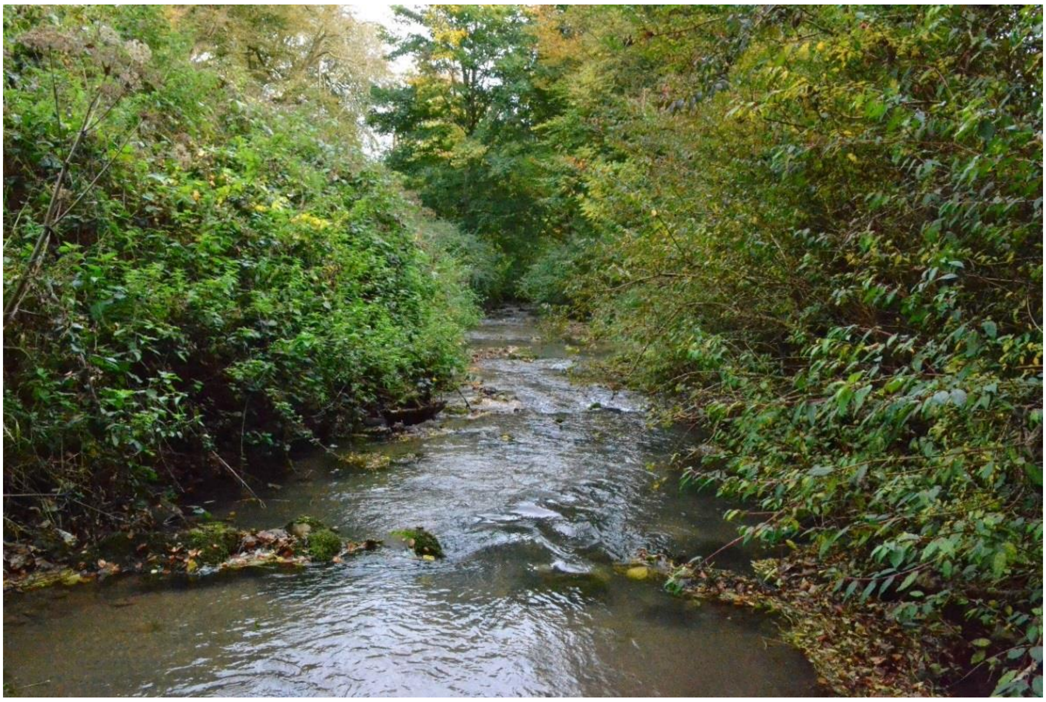


Figure H.1.3. Sample sites route map (Route 13).

Appendix I

Photographs from four Maintained, four Lost and four Gained sites

Plate I.1



a) Site 25A030100 - Ardcloony - Ballycorney Bridge – Maintained

b) Site 25A030100 - Ardcloony - Ballycorney Bridge – Maintained

Plate I.2

a) Site 25A030100 - Ardcloony - Ballycorney Bridge – Maintained

b) Site 35G020200 – River Glenaniff - Bridge u/s Lough Melvin – Maintained

Plate I.3

a) Site 35G020200 – River Glenaniff - Bridge u/s Lough Melvin - Maintained

b) Site 35C030200 - Cashel Stream (Bonet) - Bridge W. of Corratimore - Maintained

Plate I.4

a) 32C050050 - Carrowbeg (Westport) - Cloghan Bridge - Maintained

b) Extensive Algal growth at site 32C050050 - Carrowbeg (Westport) - Cloghan Bridge - Maintained

Plate I.4

a) Site 32C050050 - Carrowbeg (Westport) - Cloghan Bridge - Maintained

Plate I.5

a) Site 25W010300 - Woodford (Galway) - Rossmore Br - Lost

b) Site 25W010300 - Woodford (Galway) - Rossmore Br – Lost

Plate I.6

a) Site 25W010300 - Woodford (Galway) - Rossmore Br – Lost – Up stream of sample site.

b) Site 26I030400 - Island - Castlerea Bridge – Ballymoe - Lost

Plate I.7

a) Site 26I030400 - Island - Castlerea Bridge – Ballymoe - Lost

b) Site 30N010100 – River Nanny (TUAM) - Bridge S. of Oakmount – Lost

Plate I.8

a) Site 30N010100 – River Nanny (TUAM) - Bridge S. of Oakmount – Lost

b) Site 33A020100 - Altnabrocky - Just u/s Owenmore River confluence – Lost

Plate I.9

a) Site 33A020100 - Altnabrocky - Just u/s Owenmore River confluence - Lost

b) Extensive Algal growth - Site 33A020100 - Altnabrocky - Just u/s Owenmore River confluence – Lost.

Plate I.10

a) Site 25B100100 – Bow River - Bow River Bridge – Gained

b) Site 25B100100 – Bow River - Bow River Bridge – Gained

Plate I.11

a) Substrate at Site 25B100100 – Bow River - Bow River Bridge – Gained

b) Site 25C030100 – Cappagh River (Galway) - Metal Bridge – Gained

Plate I.12

a) Site 25C030100 – Cappagh River (Galway) - Metal Bridge – Gained

b) Site 25C030100 – Cappagh River (Galway) - Metal Bridge – Gained

Plate I.13

a) Site 34C050030 – Clydagh River (Castlebar) – Br. NW Ardarney – Gained

b) Substrate at Site 34C050030 – Clydagh River (Castlebar) – Br. NW Ardarney - Gained

Plate I.14

a) Site 34M020100 - MOY - Bridge S.E. of Cloonacool – Gained

b) Site 34M020100 - MOY - Bridge S.E. of Cloonacool – Gained

Plate I.15

a) Site 34M020100 - MOY - Bridge S.E. of Cloonacool – Gained

References – Chapter 1: High status water-bodies in the European Union

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